INVESTIGATION ON VIRUS REMOVAL BY FILTRATION

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N. SRIRAMULU

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CERTIFICATE

Certified that the work presented in this thesis entitled "Investigation on Virus Removal by Filtration" by Sri N. Sriramulu has been carried out under my supervision and it has not been submitted elsewhere for a degree.

February, 1975

Malay Chaudhuri
Assistant Professor
Department of Civil Engineerin
Indian Institute of Technolog
Kanpur

INVESTIGATION ON VIRUS REMOVAL BY FILTRATION

N. Sriramulu
Environmental Engineering Division
Department of Civil Engineering
Indian Institute of Technology, Kanpur, 1975

Studies reported in the literature on the removal of viruses from water by granular filtration were mainly concerned with the gross removal efficiencies. The objective of the present research was to study in greater detail various parameters affecting virus removal by filtration so as to be able to investigate the removal mechanisms involved. Batch sorption tests and filtration studies were conducted in the laboratory using bacteriophage MS2 against <u>Escherichia coli</u> as the model virus and silica sand and Giridih bituminous coal as filter media. Since in a natural system a fraction of the viruses may remain attached to the suspended particles constituting turbidity whereas the other fraction may remain free, the present study was conducted in two phases, viz., filtration of discrete virus particles (turbidity-free water) as well as virus in association with clay turbidity. It was thought that this would help investigate the mechanisms involved in the removal of viruses and presumably other submicron particles and contribute to our knowledge on virus removal by filtration under field condition as well.

Batch sorption experiments at pH values 6.0, 6.8, and 8.4 and ionic strength 0.02 indicated higher sorptive capacity of the Giridih coal for MS2 compared to sand. IR spectral studies also showed more active functional groups on the coal surface.

Filtration experiments with the model virus in turbidity-free water showed poor virus removal for both 0.5 mm sand (22.75 percent) and 1.0 mm coal (30.75 percent) at 4.9 m/hr. However, the removal efficiency doubled (61.5 percent) when 0.5 mm coal was used. Also, when the filter beds were treated with alum floc and cationic polyelectrolyte, the virus removal efficiency of the sand filter increased to 71.8 and 69.4 percent, respectively and the corresponding values for the coal filter were 75.2 and 4 percent. Addition of polyvalent cations into the influent also showed improvement in both untreated sand and coal. Furthermore, it was observed with untreated coal and sand that virus removal decreased at higher filtration rates and increased with an increase in temperature and decrease in pH value of the influent. Analysis of the data indicated that removal of discrete virus particles by filtration is governed by transport as well as attachment steps. The dominant mechanisms for removal are diffusion and electrokine phenomena. However, diffusion becomes more predominant at higher temperatures.

Significant improvement in virus removal was observed during filtration of virus and clay turbidity. For untreated and cationic polyelectrolyte treated 0.5 mm sand filter (4.9 m/hr), virus removals were in the range 73.0 to 87.4 and 85.5 to 92.1 percent, respectively whereas turbidity removals in the range 82.5 to 97.4 and 91.9 to 96.9 percent were observed for untreated and treated sand, respectively. The corresponding virus and turbidity removals were in the range 64.3 to 75.6 and 71.3 to 87.1 percent and 84.1 to 89.6 and 77.3 to 89.6 percent, respectively for untreated and treated dual-media coal-sand filter (9.8 m/hr). Virus and turbidity removal efficiencies were observed to be much higher following coagulation and settling. Virus removals were in the range 93.2 to 99.7 and 90.6 to 98.5 percent, respectively for single-media sand (4.9 m/hr) and dual-media coal-sand (9.8 m/hr) filters. The corresponding turbidity removals were in the range 98.0 to 99.9 and 91.5 to 97.5 percent. The results of these experiments also showed that most of the removal occur in the top layers of the filter. Analysis of the data indicated that the dominant mechanisms leading to the removal of the particles causing turbidity and the viruses in association with such particles are mechanical straining and gravity settling. However, for coagulated and settled water, the significance of gravity settling may increase compared to when such pretreatment is not used.

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1. INTRODUCTION

Virus disease transmission through water has been generally recognized. About 100 virus types have been recovered on one or more occasions from human feces (Committee Report, 1970). Many of these viruses have been subsequently found in domestic wastewaters and secondary effluents and consequently traced into raw water supplies. In addition, there has been some evidence of transport of human enteric viruses from potential reservoirs in animal pets via storm water run-off (Kasel et al., 1963). Virus contamination of ground waters is also not uncommon (Drewry and Eliassen, 1968). These are of significance in view of the report that one cell culture infective dose is sufficient to infect a man (Plotkin and Katz, 1966). Due to everincreasing demands on water resources as a result of population growth and accompanying urbanization, the period between the 'use and reuse' has been considerably narrowed down resulting in insufficient time available for significant natural inactivation or reduction of viruses and other pollutants. Therefore, water treatment plants are to be designed and operated for their maximum removal.

Of the various unit processes in a water treatment plant, the filter is of paramount importance in view of its observed role in the overall removal of colloidal particles (Table 1).

TABLE 1

REMOVAL OF E. coli AND VIRUSES BY FLOCCULATION AND FILTRATION (Abstracted after Gilcreas and Kelly, 1955)

Process	Percent Particles Passing Through		
	E. coli	Bacterial Virus	Coxsackie Virus
Coagulation-Flocculation and Settling	60	15	10
Coagulation-Flocculation and Filtration	10	2	2

It is clear that filtration after coagulation-flocculation reduces the colloidal particles concentration in the effluent 5 to 7.5 times indicating the vital role of a filter in a conventional water treatment plant.

Throughout the world there is a need today for augmenting drinking water supplies. It can be done either by increasing the number of filter units operating at normal flow rate of 4.9 m/hr (2 gpm/sq ft) or by adopting higher filtration rates using dual or multi media filters. According to the American Water Works Association Handbook of Public Water Supplies (1971), 'there is, however, no doubt at all that the dual-medium system is superior to any single-medium bed in capability to produce longer filter runs and perhaps higher quality water. The evidence is so clear that there is little justification for the use of single-medium beds.'

Very few studies on virus removal by filtration have been reported in the recent literature. These few studies were mainly concerned with gross removal efficiencies and were of 'fact finding' nature. Very little attempts were made to probe into the mechanisms involved in such removals. The reported removals varied on a wide range and were observed to be inconsistent.

Filtration process is recognized now as a two step process involving the transport of the particle to the vicinity of filter media and its subsequent attachment to the media surface. Virus particles in waters may occur in a free state as discrete particles or may be attached to other particulates. Therefore, removal of viruses in filtration presumably consists in the transport and the attachment of both discrete and attached virus particles.

The present work was undertaken in order to delineate the basic mechanisms involved in the removal of viruses by filtration. Apart from sand, bituminous coal was also used in this study with the prime objective of studying its particle retention capacity and evaluating its suitability in dual-media filtration. Various parameters that affect the process were also studied. It is believed that the results obtained from this investigation will contribute fundamental knowledge regarding the removal of viruses and other submicron particles by filtration in addition to providing pertinent information for better design and operation of filters.

2. LITERATURE REVIEW

2.1 Viruses

2.1.1 Physical and Chemical Properties

Viruses are the smallest biological form capable of producing diseases in human and in other living species. They are obligate intracellular parasites and consist of either a ribonucleic acid (RNA) or deoxyribonucleic acid (DNA) core surrounded by a protein coat. Because of the protein coat viruses exhibit an amphoteric behaviour when suspended in aqueous media and the net charge depends on the pH of the suspending medium. Isoelectric point for most viruses being below 4.5, virus particles have a net negative charge at pH values of most natural waters and wastewaters.

Viruses occur in a variety of sizes and shapes. They range from 10 to 300 nm in size and may be spherical, cubical, helical, brick-shaped or filamentous in shape. Viruses of the same type are characterized by a definite size and shape. Most of the enteric viruses of concern are about 30 nm in diameter.

Viruses can change the life process of a host cell, modify its growth rate, or even cause death. Viruses can infect a cell and remain dormant without causing any recognizable effect but the possibility of becoming virulent later exists.

2.1.2 Water-borne Enteric Viruses and Diseases Associated with Them

As has been rightly reported by Mosley (1966) and Chang (1968), any virus that is excreted in the feces may theoretically be transmitted through water. However, apart from theoretical considerations there are very few viruses for which epidemiological evidence suggests transmission by drinking water. Viruses considered as potential water pollutants are the enteric viruses which grow in or near the intestinal wall and are discharged in large numbers in feces. Chang (1970) considered the entero viruses (poliovirus, coxsackievirus, and echovirus), infectious hepatitis virus(es), the adenovirus and the reovirus as the main enteric viruses of concern. Table 2 lists the major known viruses together with the diseases with which they have been associated.

With respect to poliomyelitis, many attempts have been made to implicate drinking water as mode of transmission; however, most investigators remain unconvinced (Mosley, 1966). A vast majority of poliomyelitis outbreaks do not appear to satisfy the epidemiological criteria for water-borne transmission even though such outbreaks have been suspected from wastewater contaminated water supplies. Nevertheless, there still remains the possibility that under unusual circumstances, poliomyelitis could break out as a water-borne infection (Clarke and Chang, 1959).

TABLE 2

ENTERIC VIRUSES AND DISEASES WITH WHICH THEY HAVE BEEN CLOSELY ASSOCIATED
(After Berg, 1966)

Virus[φ]	Paralytic Poliomyelitis	Aseptic Meningitis	Pleurodynia	Herpangina	Respiratory Illnesses	Enteritis	Rash Diseases	Acute Infantile Myocarditis	Jaundice
Polioviruses	X	X	-	-	-	-	-	-	-
Coxsackieviruses group A	-	X	-	X	-	-	-	-	-
Coxsackieviruses group B	-	X	X	-	-	-	-	X	-
Echoviruses	-	X	-	-	X	X	X	-	-
Adenoviruses	-	-	-	-	X	-	-	-	-
Reoviruses	-	-	-	-	X	X	-	-	-
Infectious hepatitis virus(es)	-	-	-	-	-	-	-	-	X

(Column heads are grouped under "Disease".)

[φ]Only certain strains within the designated groups have been proven to be responsible for the designated diseases.

During recent years, infectious hepatitis has received increasing attention. Today, it is generally accepted by epidemiologists to be the only disease caused by an agent, believed to be a virus, for which evidence of water-borne transmission exists (Mosley, 1966). However, it should be noted that the hepatitis agent(s) has yet to be isolated. Drinking water has been identified as the mode of transmission in 50 known epidemics of infectious hepatitis among which the Delhi epidemic of 1955-56 is the most prominent (Mosley, 1966).

There have been no water-borne epidemics of gastroenteritis or diarrhea for which definite evidence of a viral etiology has been demonstrated. Although the evidence is scanty, it should be assumed that enteric viruses and other possible causative agents of viral gastroenteritis can be transmitted by drinking water (Committee Report, 1970).

Berg (1966) observed that over the years, the main concern had been exclusively with viruses of human origin, and there is no proof available that viruses of nonhuman origin cannot enter into and infect human cells. There has been some evidence of induced human infection by exposure to other warm blooded animals (Kasel _et al_., 1963) making another factor in the public health risk from virus pollution of water supplies. However, there has not yet been much concern about animal, plant, fish and insect viruses or

the viruses of the lower forms of life as causative agents of human diseases.

Plotkin and Katz (1966) reported that one cell culture infective dose is sufficient to infect a man. This indicates that if viruses are isolated from water that is consumed by man, there is sufficient virus in that water to infect a proportion of those that consume the water. However, the epidemiological data available so far are insufficient to give a measure of what this proportion is and it is interesting to note that in Paris, France, viruses were readily isolated from tap water with no evidence of corresponding outbreaks of virus diseases (Coin _et al_., 1965). A careful scrutiny of the data demonstrating that even very small concentrations of viruses can cause disease in man reveals that in much of this work special efforts were made to bring the viruses into contact with susceptible cells. Information available on the aspects of minimal infective dose and what portion of the infected hosts get disease is far from satisfaction.

2.1.3 Detection of Viruses in Natural Waters and Their Densities

Studies involving the detection of viruses in the water environment and the evaluation of treatment methods in removing viruses from water and wastewater have been seriously limited due to the lack of suitable quantitative

methods of sampling and concentration. Low densities of viruses in natural waters require that large quantities of water be collected and subsequently concentrated before applying conventional virological detection and isolation techniques.

Sampling flowing water or large bodies of water for viruses remains a challenge to the virologists. The basic sampling methods in use are the grab sample method and the gauze pad method (Berg, 1966). In the gauze pad method, pads of gauze or sanitary napkins are suspended in flowing waters for several days. The pads are squeezed to remove the fluid which is then concentrated and subsequently assayed for virus density. Both methods have associated advantages and disadvantages. To provide a quantitative and fully reliable sampling procedure, existing and new techniques need to be researched and developed (Chaudhuri and Engelbrecht, 1974).

Various methods for concentrating large volumes of water containing viruses into small volumes have been reported. Older methods include chemical precipitation, ion exchange, high speed centrifugation, and combination of these (Clarke et al., 1959). Recent methods studied include hydro-extraction and two-phase separation, electrophoresis, continuous-flow ultracentrifugation, passive hemagglutination, concentration with polyethylene glycol, use of soluble ultrafilters, aluminum phosphate and aluminum hydroxide precipitation and adsorption on membrane filters (Berg, 1966). Electro-osmosis and

forced-flow electrophoresis (Sweet et al., 1971), freeze concentration (Rubenstein et al., 1971), osmotic ultrafiltration (Sweet et al., 1971a), and adsorption of viruses on magnetic iron oxide (Rao et al., 1968) and insoluble polyelectrolyte (Wallis et al., 1970 and 1971) have also been studied. The effectiveness and reliability of these methods require detailed evaluation.

The gauze pad procedure is the recommended procedure for qualitative studies and the membrane filter procedure of Cliver (1966) is tentatively recommended for quantitative studies (Standard Methods, 1971). In the membrane filter procedure, volumes of water 100 gallons and more, may be passed under suction or light pressure through special types of membrane filters (0.45 µm) to which viruses adsorb. Turbid waters can be cleared by filtration through coarse pre-filters with relatively little loss of virus. Elution of the adsorbed virus is achieved by immersing the filters in 3 percent beef extract.

Recent research has led to the development of a portable virus concentrator for testing water in the field (Wallis et al., 1972). The apparatus essentially consists of a set of textile fibers to remove the nonviral components, and cellulose acetate membrane layers which adsorb viruses with subsequent elution of virus into small volumes suitable for assay. It is capable of processing raw water at the rate of 300 gallons per hour, with total virus removal from the

water and with 80 percent elution of the virus from the adsorbent. An apparatus for conditioning unlimited quantities of finished waters for enteric virus detection has also been developed recently which uses nitrocellulose membranes or epoxy-fiber glass filters as virus-adsorbent (Hill et al., 1974). In quantitative virus recovery studies using poliovirus type 1, recoveries have ranged from 25 to 50 percent with virus inputs of 16 to 50 PFU/gallon with 100 gallons samples.

Regarding the density of human enteric viruses in domestic wastewater, it was observed that their concentration varied seasonally, reaching a peak during the late summer and early fall (Gelfand, 1961). The average enteric virus density in raw domestic wastewater was estimated by Clarke et al., (1964) at 700 virus units per 100 ml. From the data available in the literature it appears that the average number in domestic wastewater is probably about 600 $TCID_{50}$ per 100 ml during the warm months and 5 $TCID_{50}$ per 100 ml during the cold months (Grabow, 1968). However, density as high as 10,000 $TCID_{50}$ per 100 ml has also been reported (Lund et al., 1969). A secondary effluent might be expected to contain 7 to 70 virus units per liter if no reduction is taken for primary treatment, 90 percent for activated sludge and 90 to 99 percent for chlorination, and if the initial enteric virus density is about 700 virus units per 100 ml (Sproul et al., 1967). Taking an initial density of 600 $TCID_{50}$ per 100 ml, 6 to 60 $TCID_{50}$ per liter can be expected in secondary effluents.

This has also been confirmed by Lund et al. (1969) from actual plant data.

Regarding surface waters the safe conclusion is that viruses are present in such waters and that failure to isolate them results presumably from their low concentrations, and the limitations of sampling and concentration procedures employed (Committee Report, 1970). Clarke et al. (1964) estimated an enteric virus density of 10 PFU/l in polluted surface waters. The reported values are 0.5 to 10 PFU/l in the River Thames (Hill et al., 1971), 13 PFU/l in the Jordan River (Committee Report, 1970), and 45 to 286 PFU/gallon in a Texas stream (Grinstein, 1970).

2.2 Virus Removal by Conventional Wastewater Treatment Processes

Virus removal in the strict sense of the term is the physical capture of the virus particle making the suspending fluid free of this particle. This is what is generally achieved in some of the physico-chemical processes such as adsorption and filtration. Virus inactivation on the other hand is making the virus particle incapable of infecting its host even though the virus particle may still retain its physical identity. Temperature effects, chemical disinfectants, ultraviolet irradiation, pH effects of the suspending medium etc. can inactivate virus particles. The action of predators on viruses can be considered either under removal or inactivation. The net result of the removal and the inactivation is

the same, i.e. making the virus particle incapable of reaching or infecting the host. The reduction in viruses during various phases of wastewater handling are briefly discussed below.

2.2.1 Storage

Virus removal due to storage is either due to natural inactivation or sedimentation. However, the storage times normally available in wastewater treatment do not permit the sedimentation of the discrete virus particle and the reduction is mainly due to the removal of virus particles adsorbed onto bigger particulates that settle. The time required for 99.9 percent reduction of viruses in natural water and wastewater is shown in Table 3.

A critical examination of table 3 indicates that virus loss depends upon (a) temperature, (b) time of storage, (c) virus species and (d) degree of pollution. The more rapid reduction of virus at higher temperatures is consistent with the general concept of a chemical process. Batch laboratory studies by Clarke et al. (1961) indicated that 1/3 to 2/3 of the polioviruses added remained in the effluents after 24 hr of settling although all but 25 percent of the suspended solids had settled.

TABLE 3

EFFECT OF STORAGE ON VIRUS SURVIVAL
(Abstracted from Clarke et al. 1964)

Virus	Days Required for 99.9 Percent Reduction of Viruses)								
	Little Miami River Water			Ohio River Water			Domestic Wastewater		
	4°C	20°C	28°C	4°C	20°C	28°C	4°C	20°C	28°C
Polio 1	27	20	17	19	13	11	110	23	17
Echo 7	26	16	12	15	7	5	130	41	28
Echo 12	33	12	5	19	5	3	60	32	20
Coxsackie A9	10	8	8	20	8	5	12	-	6

2.2.2 Activated Sludge

A number of studies have indicated that the activated sludge process is capable of removing viruses to the extent of 90 percent or more (Committee Report, 1970). Experimental data of Clarke et al. (1961) shows in general that lower volatile solids concentration, poor sludge formation and consequently poor settling resulted in less removal of viruses (Table 4). It was also observed that 60 to 75 percent removals occurred when a mixture of sewage and polio virus was aerated for 6 to 7 hours. These indicate that adsorption onto the suspended and colloidal materials is a prime factor in virus removal in the activated sludge process. Bush and Isherwood (1966) found that higher sludge volume indices significantly

TABLE 4

LABORATORY STUDY ON REMOVAL OF VIRUSES BY ACTIVATED SLUDGE AT ROOM TEMPERATURE (23-27°C)
(Abstracted from Clarke et al. 1961)

Virus	Experiment Number	Average Retention Time (hr)	Virus Concentration (PFU/ml) Initial	Virus Concentration (PFU/ml) Effluent	Percent Virus Reduction	Volatile Solids (mg/l)
	1		166,000	2,200	98.8	600
	2		509,000	20,000	96.1	650
	3		167,000	1,400	99.2	1,000
	4	7	320,000	2,800	99.1	1,100
	5		321,000	8,400	97.4	1,500
	6		314,000	2,000	99.4	1,500
Coxsackie A9						
	7		67,000	14,000	79.0	200
	8		68,000	8,000	88.0	400
	9		64,000	6,200	90.0	600
	10	6.7	53,000	5,000	91.0	600
	11		100,000	8,000	92.0	1,200
	12		40,000	3,600	91.0	1,200
	13		56,000	3,600	94.0	4,000

increased the removal of coxsackieviruses. Kelly *et al.* (1961) also reported that there were certain bacteria in the activated sludge which displayed antagonistic activity to viruses. Recently, Cliver and Herrmann (1972) showed antiviral activity of *Bacillus subtilis* and *Pseudomonas aeruginosa* against coxsackievirus A-9.

2.2.3 Trickling Filter

Data from a few field studies are available on the removal of viruses by trickling filters and no laboratory studies have been reported. Kelly and Sanderson (1959) reported that the plaque counts were reduced by about 40 percent by trickling filtration. Bush and Isherwood (1966) using the gauze pad technique with Coxsackie A-13 virus did not find any change in the number of suckling mice that survived when they were given the influent and effluent of a trickling filter. However, they stated that this should not be interpreted as indicating that no removal occurred in the filter as the influent virus concentration was sufficiently large.

A critical review of the literature indicates that virus removal by trickling filters is neither large nor consistent whereas the activated sludge process is capable of removals in the order of 90 percent or more. Eventhough there are differences in the construction, operation and maintenance of these two processes, essentially they are expected to have a similar environment to handle the pollutional load with a

probable difference in microbial flora. Whether this difference in the microbial flora is responsible to a major extent for the vast difference in removing viruses is an area still open for research.

2.2.4 Oxidation Ponds

A review of the published data indicates that oxidation ponds are effective in reducing the virus levels of raw wastewater and treated effluents (Chaudhuri, 1973). The oxidation ponds at Santee, California with a nominal detention period of 30 days gave 16 percent positive results for viruses in 63 effluent samples tested (Merrel and Katko, 1966). Effluent from an activated sludge unit with 96 percent positive results for viruses out of the 52 samples tested was the influent to the pond. While greater than 90 percent removal was reported by Arceivala et al. (1971), Nupen (1970) reported that viruses could not be recovered in the effluent when the influent contained 240 PFU/l. Presence of algae and bacteria, chemical and organic contents of the pond water, detention time and temperature, oxygen supersaturation, and sunlight all may play significant roles in the removal or inactivation of viruses in oxidation ponds (Chaudhuri, 1973). The virus removal potential of oxidation ponds, a low-cost waste treatment method in the reach of most of the developing countries, needs to be studied more extensively and more

research is needed to elucidate the specific factors responsible for virus removal or destruction.

Table 5 indicates the probable virus removals by an elaborate hypothetical wastewater treatment system. The assigned zero removal by primary sedimentation is unwarranted as the heavier solids with the adsorbed viruses settle down. Kelly and Sanderson (1959) and Mack et al. (1962) have reported isolation of viruses from the final effluents of sewage treatment plants, even after chlorination. It is evident, therefore, that the final responsibility for the safety of potable waters must rest with water treatment system.

2.3 Virus Removal by Conventional Water Treatment Processes

It is generally agreed that the most desirable way to obtain factual information regarding the efficacy of water treatment processes for removing viruses would be to examine both the raw and finished water for viral agents. However, because of a lack of a simple reliable technique to quantitatively detect a small number of viruses in large volumes of water, this approach has not been entirely feasible. A recent study of viruses in drinking water in ten water treatment plants in the United States conducted by Clarke et al. (1974) using the most advanced virus sampling and concentration methods, failed to draw any definite conclusions on the potential problem of water-borne virus disease. Most of the

TABLE 5

EXPECTED VIRUS CONCENTRATION IN EFFLUENTS FROM COMMONLY USED WASTEWATER TREATMENT PLANTS
(After Sproul, 1973)

Treatment	Removal Expected Percent	Virus in Effluent[φ] PFU/liter
Primary Treatment		
Primary Sedimentation	0	
Chlorination	50	3,500
Secondary Treatment		
Trickling Filters	50	
Chlorination	50	1,750
Activated Sludge	90	
Chlorination	50	350
Tertiary Treatment		
Precipitation of Phosphate and Sedimentation	90	
Activated Carbon Adsorption	10	
Chlorination	99	6

[φ]Assuming 7,000 PFU/l in the influent.

data on virus removal by water treatment processes reported in the literature were obtained in laboratory systems using model viruses.

2.3.1 Coagulation-Flocculation Followed by Sedimentation

The earlier studies in this area were concerned with gross removal efficiencies and relationships between virus removal and alum dose and settling time without any attempt to delineate the basic physico-chemical mechanism involved. Using the virus of the mouse adopted strain of human poliomyelitis Carlson et al. (1942) reported that coagulation-flocculation with 100 mg/l alum did not render the water noninfective. Kempf (1942) also conducted similar experiments using monkey adopted strain of human poliomyelitis and reported cent percent removal at high alum doses.

Chaudhuri and Engelbrecht (1970) presented experimental evidence that the mechanism of virus removal by coagulation-flocculation proceeds by the initial formation of a coordination complex between virus (T4 and MS2 phages) and the metal coagulant in which aluminum coordinates with carboxyl groups of the virus coat protein. This is followed by incorporation of the complex into the precipitate. These findings are in agreement with the hypothesis of formation of a metal cation (coagulant) - protein (virus) complex followed by precipitation proposed by Chang et al. (1958) and the chemical aspects of coagulation (Stumm and Morgan, 1962).

Chang *et al*. (1958) have reported some very interesting results on removal of viruses during coagulation. The experiments were designed to compare the behaviour of a bacterial virus against *Mycrococcus pyogenes* var. *albus* and coxsackievirus (as a representative of pathogenic viruses) in coagulation-flocculation and to have an estimate of the possible removals. The abstracted results of this study are presented in Table 6 along with the results of Chaudhuri and Englebrecht (1970), and Thorup *et al*. (1970). The results of Chang *et al*. indicated that increased doses of alum had an effect on the removal of both the viruses. Eventhough it was reported that the coxsackievirus was less removed compared to the bacterial virus, a critical examination of the results does not substantiate this and the differences are not very significant for practical purposes. Also it is to be noted that the concentrations of the coxsackieviruses were determined by constructing dilution mortality curves whereas the bacterial virus was estimated by the more accurate plaque count technique.

Another interesting aspect of the research of Chang *et al*. was the finding that in the presence of a bicarbonate buffer the effect of pH was less critical than in the presence of a phosphate buffer in the removal of viruses from water by alum flocculation. Also, with the same alum dose, the removals in the presence of phosphate buffer were less. No reason for this was cited. Probably the phosphate ion has an equal or similar affinity for aluminium ion as the virus particle and

TABLE 6

VIRUS REMOVAL BY COAGULATION WITH ALUMINIUM SULPHATE

Initial Turbidity of Suspension	Alum mg/l	pH	Percent Virus Removal					Percent Turbidity Removal	Remarks
			Cox-sackie	Bacterial Virus$^{\varphi}$	Polio	MS2	T4		
SiO_2 suspension added but turbidity not given in practical units	40		86.3	93.5					
	60		95.5	-					
	80	6.2	97.1	99.1					
	100		98.7	99.8					
	120		-	99.8					Chang et al. (1958)
	80	5.5	94.6	85.0					
		6.2	97.1	99.1					
		7.2	99.0	97.3					
		8.2	-	80.0					
50 mg/l	10	6.8			86			96	Thorup et al. (1970)
120 mg/l (12.5 FTU)	25.7	6.0				99.8		98	Chaudhuri and Engelbrecht (1970)
120 mg/l (12.5 FTU)	25.7	5.24					98	99	-do-

$^{\varphi}$Bacteriophage against *Micrococcus pyogenes* var. *albus*.

the aluminium available for virus was less. This has a practical significance in the tertiary treatment of wastewaters where considerable amounts of phosphates may be present. A trend of increasing virus removal with decreasing pH to 5.0 was found at Al:PO_4 ratios of 1.0 and 1.5 by Brunner and Sproul (1970) in their studies on removal of poliovirus during precipitation of phosphate from domestic wastewater with alum. For an initial virus titer of approximately 10^7 PFU/ml, the removals observed were 93 and 97 percent at pH 7.0 and 5.0, respectively.

Cationic polyelectrolytes were found effective in removing viruses from water both as prime coagulants and coagulant aids (Chaudhuri and Engelbrecht, 1971, and Thorup et al., 1970). It was also demonstrated that the concentration of cations in the water affected quite markedly the extent of virus removal when polyelectrolytes were used.

The removals achieved by coagulation-flocculation in laboratory experiments were very significant and usually greater than 90 percent some times attaining 99 percent and above (Table 6). Eventhough it is probable that there were differences in stirring rates and settling times in these experiments the removals were always above 90 percent. However, while extrapolating those data to field units proper caution must be exercised as the ideal stirring and settling conditions available in the laboratory may not be attainble

in the field. The coagulation-flocculation studies using laboratory cultures of model viruses employed virus concentrations far in excess of what is believed to occur in natural water systems. Therefore, interpretation of these data in terms of naturally occurring viruses must also be done with caution.

2.3.2 Filtration

In a review of the literature on virus removal or inactivation by water treatment processes, Sproul (1971) observed that 'very little work on virus removal by filtration at rapid sand filter rates has been published in the last eight years. Yet over the same period this process in some plants has become the only treatment unit performing the roles of flocculation chamber, sedimentation tank, and filter.'

A few observations have been made regarding virus removal by filtration through sand and garden soil. Most of the early investigations on virus removal by filtration, as reviewed by Clarke and Chang (1959), showed poor virus removal in laboratory studies in which virus suspensions were usually prepared from infected animals (Table 7). Gilcreas and Kelly (1955) showed that the efficiency of virus removal was influenced significantly by the filtration rate and alum flocculation prior to filtration improved the removal efficienc They as well as Carlson et al. (1942) observed improved virus removal efficiency when the filters were impregnated with alum

TABLE 7

VIRUS REMOVAL BY FILTRATION
(After Amirhor, 1974)

Investigators and System	Flow Rate	Virus Used	Virus Removed Percent or as Noted
Carlson *et al.* (1942)			
Rapid Sand Filtration	2 gpm/sq ft	Poliomyelitis (Pathogen for Mice)	Poor
Impregnated Filter	1.3 gpm/sq ft		Good
Kempf *et al.* (1942)			
Flocculation and Rapid Sand Filtration	2 gpm/sq ft	Poliomyelitis (Strain DG)	Poor
Gilcreas and Kelly (1955)			
Percolation Through 3 ft Soil		Coxsackie	50
		T4	20
Sand Filtration	0.2 gpm/sq ft	Coxsackie and T4	99
	2 gpm/sq ft	Coxsackie	10
Flocculation and Rapid Sand Filtration		T4	40
		Coxsackie	90
Impregnated Rapid Sand Filter		T4	99
		Coxsackie	90
Robeck *et al.* (1962)			
Sand Filtration - Slow Rate	.035 gpm/sq ft	Poliovirus Type 1	22 - 96
Sand Filtration - Rapid Rate	2 - 6 gpm/sq ft		1 - 50
With Flocculation			90 - 99
With Flocculation and Settling			99.7

floc. However, Chang et al. (1958) obtained very little, if any, virus removal when the virus suspension was added one min after alum dosing. The true effectiveness of rapid sand filters, impregnated with preformed alum floc remained to be determined.

A comprehensive study was conducted by Robeck et al. (1962) to investigate the removal of poliovirus by sand filtration (Table 7). Using 2 ft of coarse or uniform fine sand, they observed 1 to 50 percent removal of the added poliovirus at 2 to 6 gpm/sq ft (4.9 to 14.7 m/hr) and 22 to 96 percent removal at 0.035 gpm/sq ft (0.086 m/hr). Flocculation (5 to 10 mg/l alum) prior to rapid sand filtration produced 90 to 99 percent virus removal, but floc break-through accompanied by virus penetration occurred at variable times depending upon the turbidity of the raw water, floc strength, temperature and rate of filtration. The virus penetration was observed to increase without an increase in the filter effluent turbidity when a relatively clear water was treated. Increased alum dose followed by settling prior to filtration increased the removal efficiency to over 99 percent. Filtration through 2 ft sand columns at ground water movement rates showed over 99.99 and 98 percent virus removal at 3 and 4 ft/day, respectively.

The Metropolitan Water Board, London (1970-73) in its report on the investigation on virus removal by slow sand filters reported the effects of temperature and filtration

rate on virus removal but did not offer any explanation for such effects. A comparison of the virological and bacteriological data indicated that the filters appeared to be slightly more efficient in removing poliovirus than *Escherichia coli*. However, it was observed that 'this result is rather unexpected as poliovirus is much smaller than *Escherichia coli* and would normally be expected to penetrate a filter more easily than *Escherichia coli*.'

Regarding diatomaceous earth filtration, the classical study by Neefe *et al*. (1947) on the inactivation of the virus(es) of infectious hepatitis in drinking water indicated that coagulation, settling and filtration through a diatomaceous earth filter did not eliminate or inactivate the virus(es) in the contaminated water. Recently, efficient removal of viruses from water has been achieved using diatomaceous earth coated with water soluble cationic polyelectrolytes (Chaudhuri *et al*., 1974; and Brown *et al*., 1974). However, uncoated diatomaceous earth was not found to be effective in removing viruses.

During the last few years there have been some studies dealing with virus movement through soils under ground water flow conditions (Drewry and Eliassen, 1968; Filmer and Corey, 1966; Carlson *et al*., 1968; and Young and Burbank, 1973).

The investigations so far undertaken on virus removal by filtration were directed towards quantitative

results, i.e. gross removal efficiencies. Very limited attention was given to the mechanisms of virus removal by filtration except for a study by Cookson (1970) who investigated the removal of T4 phage in packed beds of active carbon. The chemical characteristics of the system were maintained at optimum conditions for virus attachment, while the physical characteristics were varied. Virus removal was interpreted by a physical transfer model based on particle transport by diffusion in a flowing fluid.

2.4 Virus Removal by Adsorption

Adsorption (attachment) from suspension onto a solid occurs as a result of either lyophobic character of the solute relative to the particular solvent or high affinity of the solute for the solid. The affinity of the solute for the solid may be a result of either electrical attraction (exchange adsorption) or Van der Waal's attraction (physical adsorption) or chemical interaction (chemical adsorption) or a combination of these. Also many adsorption processes involving organic molecules result from specific interactions between identifiable structural elements of the sorbate and the sorbent which may be designated as specific adsorptions (Mattson et al., 1969, and Weber, 1972). Therefore, it can be clearly realised that adsorption phenomena is just the attachment step and is significant in processes such as coagulation, filtration etc. The unit operation of adsorption refers to both the transport

of the sorbate adjacent to the interface and the attachment.

As first understood and introduced into the chemical engineering field, an adsorbent is generally an extremely porous solid with large internal surfaces, its external surface comprising only a small part of the total surface. A variety of adsorbents such as a variety of clays, chars, activated carbons, gels, alumina, silicates, etc. each with a distinct affinity for adsorption of certain vapours have been developed for industrial use in the recovery of solvents, in fractionation of mixed gases etc. However, as far as the water industry is concerned the term adsorption initially referred to the processes employing adsorbents directed to the elimination or reduction of taste and odor by the use of carbon, reduction of hardness by ion exchangers such as zeolites, etc. Spalding (1930), Stern (1931), Trowbridge (1931) and Pirnie (1931) were the first few to use powdered active carbon for removing taste and odor in water supplies. Generally, powdered carbon was added in small quantities to the water and later allowed to settle or was filtered out. Harrison (1931) initiated the application of granular active carbon on a municipal scale. However, the use of granular carbon was limited by the expense of such units. About the same time, carbon much less active than many of the marketed active carbons was used for the removal of taste and odor in England. Recently, attention has been directed towards the use of coal as a potential sorbent in water and wastewater treatment. Several

investigations on the use of coal for sorption of selected pesticides and phosphorous containing compounds (King et al., 1969) as well as organics from domestic and industrial wastewaters (Johnson et al., 1965; Carlton et al., 1965; Shannon and Silveston, 1968; and Rosenkranz and Silveston, 1970) have been conducted with varying degrees of success.

Particle removal in adsorption process is conceived to consist of the following steps: (1) transport of the particles from the bulk liquid across the interface with the sorbent (film diffusion), (2) transport into the pores of the adsorbent (pore diffusion), and (3) the final adsorption or attachment onto the surface (Weber, 1972). The overall removal rate is governed by the slowest of the above steps. For processes in which the rate limiting step is adsorption (attachment) on the exterior surfaces of the adsorbent (e.g. virus removal) the rate is expected to vary as the reciprocal of the diameter of the adsorbent particles for a given weight of the adsorbent (Weber, 1972). It would, therefore, appear desirable to employ particles of as small a diameter as conditions of efficient operation allow so that high rates of adsorption may be obtained.

In batch type contact operation a quantity of carbon is mixed continuously with a specific volume of water until the pollutant in the solution has been decreased to a desired level. If finely powdered carbon is used, separation of the spent adsorbent from the water is generally difficult

and has to be filtered out. Conversely, the use of large particles which may be removed more readily after exhaustion requires long periods of contact between the solution and the adsorbent necessitating larger basins for retaining the water during treatment. Column type continuous flow operation appears to have distinct advantages over the batch type because rates of adsorption depends on the concentration of solute in the solution being treated. For column operation, the carbon is continuously in contact with a fresh solution. Consequently, the concentration in the solution in contact with a given layer of carbon in a column is relatively constant (Weber, 1972). Further, when the adsorbent is exhausted it can be restored in placc to some degree by backwashing.

2.4.1 Active Carbon

Cookson (1967 and 1969) conducted studies on the adsorption of bacteriophage T4 on powdered active carbon (2.5 μm) under agitated nonflow conditions. He attributed the removal of T4 phage to the electrostatic attraction between the amino groups on the virus and the carboxyl groups on the carbon and designated the viral tail fibers as the adsorbing units. Probably this postulation is based on the confirmed mode of T4 phage infection onto their host cells. He concluded that adsorption on carbon can be completely blocked by esterifying these surface groups eventhough he did not

present any experimental proof for this. Also, it was observed that there existed an optimum condition of pH and ionic strength (pH 7.0 and I 0.08 for T4) for maximum adsorption. He explained the difference in performance at various pH values and ionic strengths based on the probable changes in the configuration of tail fibers and tail sheaths of T4 phage.

Knowledge on the ranges of electrostatic and Van der Waal's forces indicates that they are of the same order and even in case the surface groups are blocked by esterifying, the Van der Waal's forces would be operating and, therefore, the adsorption would take place but may be to a lesser extent. For adsorption due to Van der Waal's forces the tail part of T4 phage has no specific significance.

However, Cookson's work led to the following important conclusions:

a. Kinetics of bacteriophage T4 adsorption on active carbon can be described by a reversible second order equation, first order with respect to both virus and carbon concentration.

b. Adsorption obeys Langmuir isotherm and is physical in nature.

c. The virus is completely excluded from the micro pores because of the large size of the virus particle.

Sproul <u>et al</u>. (1967 and 1971) reported results of studies on the removal of viruses (T2 phage and poliovirus

type I) added to secondary effluents by filtering through granular active carbon beds (15 cm deep). Results typical of the adsorption processes were obtained. The exhaustion point for the virus reached long before the exhaustion point for organic matter (COD). Eventhough no explanation for this finding was given, this could probably be due to the participation of micro pores in the removal of other organic molecules contributing to COD, whereas the viruses were removed only due to sorption on the exterior surface of the carbon. When the run was continued beyond the exhaustion point for the viruses, the virus concentration in the effluent was observed to be greater than that in the influent. This suggests desorption of the previously sorbed viruses and the competition due to other organic molecules (probably because of their high concentration) with viruses for sorption sites. It was also reported that the maximum sorption of the virus and the organic matter occurred early in the filter run. They did not explain this finding as well. The probable reason could be that the secondary effluent still having considerable COD required large sorption sites and shallow sorption beds might get exhausted soon. Under these circumstances, as one can expect, the virus removal efficiency was poor and did not exceed 40 percent in any case. Generally, in water filtration particle removals do not indicate this trend.

2.4.2 Coal

A detailed investigation on the sorption potential of bituminous coals for removing viruses from water has been conducted at the Environmental Engineering Laboratory, Indian Institute of Technology, Kanpur (Oza, 1974). Using MS2 and T4 phages in non-flow agitated systems, bituminous coals (Giridih and Churcha) were shown to be about 60 to 68 percent as effective as active carbon. The sorption reaction was interpreted as an interaction between the amino groups on the virus coat protein and the organic oxygen and hydrogen functional groups on coal. Effect of chemical environment on electrokinetic behaviour of coal was found to influence sorption. Virus sorption on coal was observed to be an exclusively external surface phenomenon and extraneous organic matter interfered with sorption.

The studies on virus removal by sorption on active carbon or coal indicate that the virus concentrations that occur in secondary effluents cannot be economically reduced to an acceptable level by a sorption process during tertiary treatment. Either they could be destroyed by proper chlorination or could be removed in a water treatment plant by which time the organic content of secondary effluent which interferes with sorption is diluted or reduced manyfold. The data obtained from agitated non-flow experiments of Cookson and Oza have limited direct applicability in a water treatment plant. However, in times of suspected water-borne epidemics

the use of powdered carbon or coal with turbulent mixing far ahead of filtration may prove advantageous.

2.5 Mechanisms of Filtration

Iwasaki (1937) based on his observations, proposed in mathematical terms the efficiency of removal of impurities in a water filter and the effect of clogging on its efficiency. He suggested that the change in concentration of suspended particles per unit depth in a granular filter is proportional to the local concentration and developed the following equation*:

$$-\frac{\partial C}{\partial L} = \lambda C \qquad 1$$

where, C = local concentration of particles in the flow

L = distance from the inlet face of the filter media

λ = filter coefficient which is a measure of the efficiency of the filter.

Agrawal (1966) gave theoretical justification for this equation based on the assumptions of the theory of air filtration that the total removal is contributed by individual media grains and an individual media grain totally removes all particles from a certain percentage of its cross-sectional area confronting the flow. This fraction is defined as the

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* Mathematical symbols are explained when they first appear in the text. All symbols used henceforth are arranged alphabetically in Appendix-I.

efficiency of an individual grain. It can be shown:

$$-\frac{\partial C}{\partial L} = \frac{1.5(1 - f_o)}{d_m}\eta \cdot C \qquad 2$$

and thus,

$$\lambda = \frac{1.5(1 - f_o)}{d_m}\eta \qquad 3$$

where, η = efficiency of an individual grain

f_o = porosity of clean filter

d_m = diameter of the filter grain.

From his observations Iwasaki also proposed the following equation:

$$\lambda = \lambda_o + c\sigma \qquad 4$$

where, λ_o = filter coefficient for clean filter

σ = specific deposit

c = a constant.

This equation indicates that the efficiency of a filter increases with deposition of suspended matter in the filter.

Based on practical observations that deposits initially improve filter efficiency but later increasing deposits cause the efficiency to decline, Ives (1960) proposed:

$$\lambda = \lambda_o + c\sigma - \frac{\varphi\sigma^2}{(f_o - \sigma)} \qquad 5$$

where, φ is a constant.
Ives equation directly relates the filter coefficient (λ), to two fundamental parameters, f_o and σ. The proposed constants, λ_o, c and φ would be constant only for a particular filter system and operating conditions. Ives correlated λ_o, c and φ to the parameters. filter grain diameters (d_m), absolute viscosity (μ) and filtration velocity(V_o) through three new constants K_1, K_2 and K_3, as follows:

$$\lambda_o = \frac{K_1}{d_m \ V_o \ \mu^2} , \qquad 6$$

$$c = \frac{K_2}{d_m \ V_o \ \mu^{1.2}} , \quad \text{and} \qquad 7$$

$$\varphi = \frac{K_3}{d_m \ V_o \ \mu^2} \qquad 8$$

where, V_o = superficial velocity of filtration
μ = viscosity.

K_1, K_2 and K_3 are to be determined by pilot plant studies for a particular suspension and filter media.

Ives derived Equation 5 on certain assumptions about the geometry of the deposits which have been found to be too restrictive making the equation not applicable to all types of suspensions. Later, Ives proposed a more general form of Equation 5 as follows:

$$\lambda = \lambda_o\left(1 + \frac{b\sigma}{f_o}\right)^Y \left(1 - \frac{\sigma}{f_o}\right)^Z \left(1 - \frac{\sigma}{\sigma_u}\right)^X \qquad 9$$

where, b = a geometric constant relating to the packing of the filter grains

σ_u = the ultimate or saturation value of specific deposit (< porosity f_o)

Y, Z, X = empirical exponents.

It was also shown by Ives (1971) that by suitable choice of the exponents, Y, Z and X, mathematical models of Iwasaki (1937), Ives (1960), Mackrle et al. (1965), Shekhtman (1961), Heertjes and Lerk (1967), and Maroudas (1965) could be expressed.

Also, Ives (1971) stated that Equation 9 was not based on detailed examination of the filtration mechanisms but on more general assumptions on the importance of pore geometry and interstitial velocity. His main emphasis was on physical parameters such as diameter of the media grain, viscosity of water, and the velocity of filtration. Similarly, the studies by other investigators were also mainly concerned with equating the filter performance in terms of only physical parameters and ignored the chemical and electrokinetic aspects involved. Therefore, all the mathematical models developed may predict the performance of a filter as long as the same suspension is filtered through the same filter media under the same operating conditions. O'Melia and Stumm (1967)

pointed out that these filtration models cannot predict the performance of the same filter using another suspension or vice versa.

In a review of the theories of filtration, O'Melia and Stumm (1967) suggested that the filtration process should be conceived to involve at least two distinct steps: a transport step and an attachment step. Particle transport is a physical process and is principally affected by those parameters which govern mass transfer. Particle attachment, on the other hand, is basically a chemical process and is influenced by both physical and chemical parameters.

Yao et al. (1971) introduced into the filtration equation a collision efficiency factor (α) to reflect the importance of the electrochemical characteristics of the filter system and proposed the following equation:

$$-\frac{\partial C}{\partial L} = \frac{1.5(1 - f_o)}{d_m} \alpha \eta C \qquad 10$$

where, α is defined as the ratio of the number of collisions which succeed in producing attachment to the total number of contacts that occur between the suspended particles and the filter media. α can have a maximum value of 1.

It is now well recognized that the removal of suspended particles within a filter involves at least two separate and distinct steps: a transport step which move suspended particles to the immediate vicinity of the

solid-liquid interface presented by the filter (i.e. to a grain of the media or to another particle previously retained in the bed) followed by an attachment step (O'Melia and Stumm, 1967, and Ives and Gregory, 1967). However, the ultimate retention of the suspended particles is believed to be dependent on the physical chemistry of the system, and therefore significant physicochemical variables must be considered (Boyd and Ghosh, 1974).

In rapid filtration most of the particles to be removed from suspension are smaller than the pore spaces in the filter. If particles strictly follow the fluid stream lines, many of them would never be transported to the immediate vicinity of a grain surface and removed from the flow. Therefore, mechanisms of transport that move the particles across the stream lines and adjacent to the grain surface have to be considered. The transport step will control the rate of removal of particles if attachment is favourable.

2.5.1 Transport Mechanisms

Several transport mechanisms have been considered in the literature. They probably act simultaneously but their relative importance depend on the characteristics of the suspended particles (principally size, density and shape), the characteristics of the flow (principally velocity, viscosity and temperature) and the characteristics of the

filter media (principally surface area, pore size, shape and volume) (Ives, 1970).

2.5.1.1 Mechanical Straining

This is the simplest and one of the most popular mechanisms proposed by Hazen (1904), Fair and Geyer (1954), Camp (1964), Hall (1957) and Agrawal (1966). This could be considered in fact as a complete removal mechanism and is probably not affected by variations in attachment conditions. Pore size in a filter bed has been reported to be of the same order as media grain size from microscopic observations. Therefore, the particles to be removed being in general smaller than the pores, the removals could be mainly at the narrow corners of the interstices. According to Ives (1971), straining can take place when the concentration of particles is too high in which case many particles may arrive simultaneously at a pore opening and jam in it by arching action.

Hall (1957) and later Agrawal (1966) worked out mathematical expressions for the removal at the narrow corners. The basis of such calculations is that the area of the narrow corners of crevices which would be very much smaller than the total pore area is considered effective in the removal by the straining phenomenon. Based on Hall's expression, Agrawal proposed the following equation:

$$(\lambda_o)_{St} = 3.5\, d_p^{3/2}\, d_m^{-5/2} \qquad 11$$

where, $(\lambda_o)_{St}$ = filter coefficient (for clean filter) due to straining.

In comparing the areas of the narrow corners of the crevices to the total area of the interstices and attributing this to the fraction of particles that could be removed, it is assumed that the suspended particles are uniformly distributed all over the cross-section which is far remote from the true situation. Actually, because of the laminar velocity distribution the velocity of flow would be the least at the sharp corners of the interstices, and most of the suspension would be flowing in the middle part of the pores. A simple comparison of geometrical areas for removal would therefore overestimate the contribution by this mechanism.

2.5.1.2 Gravity Settling

This has been rightly considered the most significant mechanism of transport in filtration since the interstices offer small shallow settling basins with very low surface loading. According to Ranz (1951), the efficiency of removal by an individual media grain (η) due to gravity settling is equal to the ratio of the settling velocity of the particles to the superficial velocity of fluid (V_o). Thus

$$\eta = \frac{g}{18} \frac{(\rho_p - \rho_f)}{\mu} \frac{d_p^2}{V_o} \qquad 12$$

where, g = acceleration due to gravity

ρ_p = particle density

ρ_f = fluid density.

Hence,

$$(\lambda_o)_{GS} = 1.5 \frac{(1 - f_o)}{d_m} \eta$$

$$= \frac{g}{12} \frac{(1 - f_o)(\rho_p - \rho_f)}{\mu V_o} \cdot \frac{d_p^2}{d_m} \qquad 13$$

where, $(\lambda_o)_{GS}$ = filter coefficient (for clean filter) due to gravity settling.

Agrawal (1966) expressed doubts regarding contribution by this mechanism because Ranz developed the above equation (for η) for settling aerosol particles from gas streams flowing horizontally through the filter whereas in a water filter the fluid flows in a vertical direction with a velocity several times greater than the settling velocity of the particle. However, it is felt that this mechanism greatly contributes to the transport of particles for the following reasons:

(a) stream lines approaching a filter grain have to diverge as the flow passes around it making the flow non-vertical, and

(b) although many particles may be small and have a density little more than water with consequent small Stoke's settling velocities, they may be in regions of very low fluid velocity due to the laminar flow velocity distributions in the filter pores (Ives, 1971).

The chance of a particle being deflected by gravity to a grain surface, therefore, depends on the relative magnitudes and orientation of the fluid streamline velocity vector and the gravitational velocity vector. Also, Ives (1971) referred to visual observation of particles collecting as caps on the downstream side of the filter grains even in upflow filtration.

Recently, Fitzpatrick and Spielman (1973) reported experimental and theoretical investigations describing the mechanisms underlying micron sized and larger particles collection in water filtration through deep granular beds. For such large particles in water, Brownian motion plays less significant role, and it was shown that the dimensionless filter coefficient can be correlated by dimensionless adhesion and gravity groups.

2.5.1.3 Interception

If particles remain in streamlines which approach the grain surface to within a particle radius, the particles will contact the surface. All particles transported to these boundary streamlines will owe their final contact to

interception and, therefore, could be considered as a boundary condition of all transport mechanisms. As stated by Ives (1971) this well-known concept of air filtration was introduced to water filtration by Stein (1940). It was investigated experimentally by Ison and Ives (1969) and Yao (1968) who characterized its effect (I) as follows:

$$I \quad \alpha \quad \left(\frac{d_p}{d_m}\right) \qquad 14$$

2.5.1.4 Inertia

As previously stated, streamlines approaching a filter grain have to diverge as the flow passes around it. If particles have sufficient inertia, they would maintain a trajectory which lead to their collision with the grain. Solution of the equation of motion of the particle and the fluid indicates that such a transport (N) is given by:

$$N \quad \alpha \quad \frac{\rho_p \, d_p^2 \, V_o}{\mu \, d_m} \qquad 15$$

Although this assumes significance in air filtration due to high velocity of flow, and low value of the dynamic viscosity, it has insignificant values in water filtration as shown by Ives (1960), Yao (1968), Ison (1967), and Agrawal (1966). Ison visually demonstrated the absence of collected particles on the upstream faces of grains in an upflow filter.

2.5.1.5 Diffusion

Diffusion is a very important transport mechanism for small particles like viruses. As reviewed by Ives (1971) and Agrawal (1966), Brownian motion is observed to impart a random movement to very small particles in water due to the thermal energy of the water molecules. For particles greater than about 1 μm in diameter the viscous drag of the fluid and inertia of the particle restrict this movement. In general, the number of particles moving in any direction is the same as that of particles moving in an opposite direction and there is no net movement of particles. However, if particles colliding with a filter grain were removed, a concentration gradient will be setup in the vicinity of the grain surface. Under this concentration gradient, more particles will be moving from the bulk fluid towards the filter grain than in the opposite direction. Such a net flow of particles due to Brownian motion, is called Brownian diffusion.

As shown by Einstein (1907 and 1909), Ficks's first law of motion

$$\text{flux} = - D_{BM} \frac{dc}{dy} \qquad 16$$

applies to such net flow of particles where, $\frac{dc}{dy}$ is the concentration gradient and D_{BM} is the coefficient of Brownian diffusion.

Einstein also showed:

$$D_{BM} = \frac{kT}{3\pi\ \mu d_p} \qquad 17$$

where, k = Boltzman's constant

T = Temperature, ^oK.

The temperature of water which mainly affects its viscosity has a large affect on the D_{BM} value and consequently on the transport of particles due to Brownian diffusion. Equation 16 which represents a theoretically sound expression for diffusional flow in a static medium is of limited value when applied to a medium in motion.

The transport of a particle in a moving liquid is governed by two quite different mechanisms (Levich, 1962). First, there is a molecular diffusion as a result of concentration gradients as discussed previously. Second, the particles are entrained by the moving fluid and are transported with it. The combination of these two processes is called convective diffusion of the particle in a liquid.

As demonstrated by Cleasby and Baumann (1962) the flow in all types of filters (even for clogged filters) is laminar Although changes in viscosity and density alter the Reynolds Number, the magnitude of such changes cannot be expected to alter the flow regime in any substantial way. Assuming that the liquid motion is steady and no significant

temperature and pressure gradients exist, the diffusion flux (J_D) can be expressed as:

$$J_D = - D \text{ grad } C. \qquad 18$$

The diffusion coefficient (D) is a function of both concentration (C) of the solution and temperature. However, if the concentration of the solution is low, D can be considered constant and independent of composition. In addition to the diffusional flux (J_D), a flow of matter

$$J_{conv} = CV \qquad 19$$

travels with the volume of liquid (V) where J_{conv} refers to convective flux. The total mass flux is, therefore, the sum of the convective and the diffusional fluxes and is expressed by the vector:

$$J = CV - D \text{ grad } C \qquad 20$$

If the solution is not isothermal or if it is subjected to significant external forces (e.g. a gravitational field or in the case of an ionic solution, an electrical field) additional terms will enter into the equation.

The relationship between the convective and diffusional transfer of matter can be described by a single parameter, the dimensionless Peclet Number (P_e):

$$P_e = \frac{U_o L}{D} \qquad 21$$

where, U_o = characteristic flow velocity

L = characteristic length along which the major change in concentration takes place

D = diffusion coefficient.

When the Peclet Number is small, the concentration distribution is determined largely by molecular diffusion. Such a situation occurs (for a given D) at sufficiently low liquid velocities and in regions of small dimensions. Conversely, when the Peclet Number is large, the concentration distribution is determined essentially by convective transfer and molecular diffusion can be neglected. The diffusion coefficient in liquids is so small that even at low flow velocities the mass transport by the moving liquid begins to predominate over molecular diffusion. Levich (1962) developed the following equation:

$$\eta_D = 4.04\ P_e^{-2/3} \qquad 22$$

where, η_D is the theoretical single collector efficiency due solely to the convective diffusion.

Spielman and Friedlander (1974) presented a theoretical analysis to describe the deposition of Brownian particles which interact with collectors through an attractive

repulsive potential. They showed that when the interactions were confined sufficiently near the media grain surface the process was equivalent to convective diffusion with a first order surface reaction at the grain.

2.5.1.6 Hydrodynamic Action

The flow in the filter pores is laminar with a velocity gradient, i.e. a shear field exists (Ives, 1971). In a uniform shear field a spherical particle would rotate with a consequent accompanying spherical flow field which would cause the particle to migrate across the shear field. If either the shear field is not uniform, as in the filter pores, or the particle is not perfectly spherical and also deformable, as the suspended particles in a filter influent, the particle will be deflected in an irregular, unpredictable way due to out-of-balance forces moving it across the streamlines. The net result is that the particles will exhibit an apparently random drifting motion across the streamlines which may cause them to collide with grain surfaces. The complexity of filter pore geometry defies theoretical analysis in fluid shear fields (Ives, 1971) and several alternative dimensionless numbers could be used to characterize this phenomena (Ison, 1967).

2.5.1.7 Combined Transport Mechanisms

Taking into account the interception,

diffusion, sedimentation, and hydrodynamic mechanisms as the principal transport mechanisms, Ives (1971) proposed the following equation for the fraction of suspension retained in a filter layer one grain diameter thick(ψ) :

$$\psi = \text{const.} \left(\frac{d_p}{d_m}\right)^{\alpha} \left(\frac{kT}{3\pi\mu d_p V d_m}\right)^{\beta} \left(\frac{g(\rho_s - \rho_f)d_p^2}{18\ \mu V}\right)^{\gamma} \left(\frac{\mu}{\rho_f d_m V}\right)^{\delta} \qquad 23$$

where, α, β, γ and δ are positive exponents. On collecting the terms together

$$\psi = \text{const.} \frac{d_p^{\alpha-\beta+2\gamma}}{\mu^{\beta+\gamma-\delta}\ d_m^{\alpha+\beta+\delta}\ V^{\beta+\gamma+\delta}} (kT)^{\beta} (\rho_s - \rho_f)^{\gamma}\ \rho_f^{-\delta} \qquad 24$$

This indicates, in general, that an increase in particle size d_p will improve filtration except for diffusion transported particles where β is large compared to $\alpha+2\gamma$. There is a particle size which gives the least filtration efficiency because that particle will be too large for efficient diffusion but too small for interception and sedimentation to be effective. Yao (1968) reported this size to be around 1 μm. Equation 24 also explains the general trend of results observed in filtration in relation to the change in the diameter of filter grain, velocity of filtration and temperature.

Agrawal (1966) discussed at length the quantitation of contributions by each filtration mechanism for various particle sizes and filter media. Table 8 shows the computed

TABLE 8

COMPUTED λ_o VALUES BY VARIOUS MECHANISMS (cm^{-1})

Mechanism		Diameter of the Particle (d_p)								Equation Used in Computation
		0.025 µm	0.1µm	0.5µm	1µm	2µm	5µm	10µm	20µm	
Mechanical Straining		0.0125 X10^{-2}	0.1 X10^{-2}	0.111 X10^{-1}	0.0062	0.0175	0.0693	0.196	0.56	$\lambda_o = 3.5 d_p^{3/2} d_m^{-5/2}$ Hall (1957)
Gravity Settling	$\rho_p=2$	4.09 X10^{-6}	6.55 X10^{-5}	1.64 X10^{-3}	0.0642 X10^{-1}	0.0257	0.1603	0.642	2.565	$\lambda_o = K \cdot (\frac{g}{18} \cdot \frac{(\rho_p - \rho_f) d_p^2}{\mu V_o}$
	$\rho_p=1.1$	4.09 X10^{-7}	6.55 X10^{-6}	1.64 X10^{-4}	0.0642 X10^{-2}	0.2565 X10^{-2}	0.1603 X10^{-1}	0.0642	0.2565	Agrawal (1966)
Interception		6.18 X10^{-8}	9.9 X10^{-7}	2.475 X10^{-5}	1.0 X10^{-4}	4.0 X10^{-4}	2.485 X10^{-3}	9.9 X10^{-3}	3.955 X10^{-2}	$\lambda_o = K \cdot 1.5 (\frac{d_p}{d_m})^2$ Agrawal (1966)
Diffusion		5.82 X10^{-2}	2.32 X10^{-2}	7.88 X10^{-3}	4.97 X10^{-3}	3.135 X10^{-3}	1.705 X10^{-3}	1.073 X10^{-3}	6.78 X10^{-4}	$\lambda_o = K \cdot 0.9 (\frac{kT}{\mu d_p d_m V_o})^{2/}$ Levich (1962

$K = \frac{1.5(1-f_o)}{d_m}$, $f_o = 0.45$, $d_m = 0.05$ cm, $\mu = 10^{-2}$ poises, $T = 300^oK$, $V_c = 0.137$ cm/sec,

λ_o values due to various mechanisms for 0.5 mm filter grain, which gives at a glance the contribution by each mechanism. In computing λ_o values the collision efficiency factor is assumed as unity in each case. The merits and demerits of this assumption will be discussed in the chapter on results and discussion.

2.5.2 Attachment Mechanisms

The mechanisms of attachment are not as well formulated as those of transport. Generally, it is believed that they are dependent upon electrical double layer interaction, Van der Waal's forces and mutual adsorption (Ives, 1971).

2.5.2.1 Electrical Double Layer Interaction

Between surfaces in water the interaction of electrical double layers can lead to attraction or repulsion, depending on whether the surfaces have potentials of an unlike or like sign, respectively. Since silica sand and other filter media as well as suspended particles in water possess a negative potential, the double layer interaction will usually inhibit attachment; however, because of the magnitudes of these potentials and their extremely small range of action in natural waters such electrical effects are negligible in most cases (Ives, 1971).

2.5.2.2 Van der Waal's Forces

These universal attractive forces between atoms and molecules are due to their electronic nature and are additive, leading to attraction between grain surfaces and particles in water, although they are of extremely limited range (usually less than 50 nm) (Ives, 1970 and 1971). London extended the concept of attraction between individual molecules to apply to clusters of molecules, i.e. colloids by integrating the influence of all the molecules on all the other molecules. Mackrle and Mackrle (1961) made an attempt to correlate the removal to molecular forces. They conceived a film of small thickness around each filter grain which they termed as adhesion space. All particles entering such space were assumed to be removed. They assumed the motion of a particle within the adhesion space to be governed by Van der Waal's forces similar to gravitational pull on a particle in a sedimentation tank. However, their work was criticized later because they calculated a value for the Hamaker constant that was too high by including inapplicable terms in the computation (Ives, 1971). Also, because both electrical double layer and Van der Waal's forces interact between the surfaces, over about the same range of distances, the two should be considered together.

2.5.2.3 Bridging Action

Another attachment mechanism can be due to bridge formation by polymers such as polyelectrolytes or

hydrolysis products of alum by having one end attached to the grain surface and the other to the particle. Further, Ives and Gregory (1967) and Carlson et al. (1968) observed that Ca^{++} and H^{+} can serve as links between an anionic polymer and a negative site on a surface. Mints (1969) stated that polymers bound suspended particles into aggregates, giving them increased strength, size and weight.

2.5.3 Electrokinetic Removal Mechanism

The significance of electrokinetic forces in filtration has been well demonstrated (Agrawal, 1966, and Yao et al., 1971). It is evident from the finding of Agrawal (1966) that electrokinetic forces by themselves constitute a complete removal mechanism comprising both transport and attachment as this mechanism takes into account both physical as well chemical parameters of the system. Therefore, a separate discussion of such mechanism is warranted.

According to Agrawal (1966), the effect of small electrical charges existing either on the particles or on the filter fibers on the removal efficiency was observed and analysed first in the case of air filtration. Gillespie (1955), and Kraemer and Johnstone (1955) presented a theoretical analysis of collection of aerosol particles in the presence of electrostatic fields. Such an approach may be suitable in the filtration of air and similar poorly conducting fluids when the suspended particles or filter media or both carry

surface charges; the surface charges would be discharged only on colliding with oppositely charged particle or grounded surface. Suspended particles as well as filter media in water carry small electrical charges due to selective dissociation of certain surface groups or adsorption of ions from the suspending medium. Water is a significant conductor and in addition the water in a filter is well grounded. Therefore, the maintenance of small charges carried by individual particles or filter media could be possible only by double layer phenomena, i.e., the net charge becomes zero at a short distance from the particle surface. However, Gregory (1964) on the basis of Bierman's approach calculated the change in the interaction energy for various experimental systems as a function of separation distance 'd' and concluded that the double layer interaction energy and forces may not extend too far in water. Therefore, the double layer interactions are ineffective in transporting the particles in filtration. Also, Agrawal (1966) discussed the limitation of the double layer interaction models by Bierman (1955) and Overbeek (1952) for application in water filtration. The main objection was that those models are for static conditions and therefore cannot be applicable for the kinetic water filter.

Due to the flow of water the double layers surrounding media particles are continuously sheared away setting up a net potential called streaming potential. This potential is an overall or net phenomena and not a local

effect immediately in the vicinity of the media grains and such a potential, however small, indicates an electrical field. The intensity of the electrical field will be proportional to the streaming potential developed. The electrical field established is expected to be three dimensional. The field intensity in the fluid will not be zero unlike in the double layer model where the field intensity is essentially negligible outside the thickness of the double layer (about 1 μm or less). The force acting on a particle in such a field would be proportional to the product of the field intensity and the charge on the particle. Since the field established depends upon the shearing of double layers and the shear in a laminar flow decreases from the surface to the centre line of flow, the electrokinetic forces will decrease as the particle distance from the media increases. However, it was hypothesized that even at the centre line of the pore channel which may be as much as 100 μm from the surface of the media, the force should not become insignificant (Agrawal, 1966). Further, in addition to the electrokinetic force experienced by the particle resulting from the charged nature of the media, electrokinetic forces due to shearing of double layers of other particles due to a relative motion may also be acting on the particle. The latter forces would be zero if the suspension is uniform when electrokinetic forces would be equal in all directions and will cancel out. In a filter, the suspension is not uniform throughout because

of the presence of media grains, and a concentration gradient from the bulk fluid to the collector surfaces resulting from the removal of particles. This net force will be proportional to square of the charge on each particle, concentration of particles, volume of the collector grain, and some power of the distance from the collector grain.

Agrawal (1966) developed the following equation to relate the effect of electrokinetic forces to the filter coefficient when the electrokinetic mechanism is the dominant mechanism in the removal of particles:

$$\lambda_o = \frac{7(1 - f_o)}{d_m} E \qquad 25$$

and

$$E = \left\{ \frac{K_2 \zeta_p^2 N d_m}{\mu V_o} - \frac{K_1 \zeta_m \zeta_p d_p}{\mu V_o} \right\} \qquad 26$$

where, λ_o = filter coefficient for clean filter

f_o = porosity

d_m = diameter of filter media, cm

d_p = diameter of suspended particles, cm

ζ_m = zeta potential of media

ζ_p = zeta potential of suspended particles

N = numerical concentration of particles, no/ml

μ = viscosity poise

V_o = superficial velocity of filtration, cm/sec

K_1, K_2 = constants.

It is evident from the foregoing discussion that electrokinetic phenomena play an important role in particle removal in filtration. This was recognized earlier by Jorden (1963) and Baumann and Oulman (1964 and 1970). Therefore, measurement of the zeta potentials of filter media as well as suspended particles is of considerable significance in any study attempted towards delineation of particle removal mechanisms in filtration.

2.6 Attachment Potential of Carbon and Coal, and Sand

The surface of solids may be considered as an extreme case of lattice defect. On a perfectly clean surface, there is an abrupt termination of the regular array of atoms and the coordination of the atoms must be different from that within the structure. Therefore, the atoms on the surface will have unsatisfied bonds capable of reacting with other elements or compounds. Surface groups consisting of atoms foreign to the structure are, therefore, formed and the properties of a surface are influenced by the surface groups (Boehm, 1966).

The surface charges on the filter media are due to selective dissociation of certain surface groups or adsorption of ions from the suspending fluid. Therefore the surface groups of the filter media influence the zeta-potential values and through them the electrical double layer interactions. Also the residual charges on the surface groups may form H^+

bonding or ionic bonding or covalent bonding with the particles in suspension. In case there exists a difference in the attachment potential of different types of filter media, for example sand and coal (of the same size), it can be due to differences in the surface chemistry of these materials.

2.6.1 Surface Groups on Carbon and Coal

Among the surface compounds of carbons and coals, the most important are those with oxygen and sulphur. Halogens and hydrogen, can also serve as end groups (Boehm, 1966). Carbon oxygen complexes are by far the most important in influencing surface reactions, surface behaviour, wettability, and electrical and catalytic properties of carbons. Rhead and Wheeler (1912 and 1913) concluded that the oxygen combines with a mass of carbon to form a physicochemical complex C_xO_y of variable composition which decomposes over a wide range of temperature, giving a mixture of CO and CO_2.

Besides oxygen, hydrogen is also present in most of carbons as could rightly be expected because carbons are pyrolysed residues of organic compounds. The hydrogen is present as chemisorbed water and as phenolic, hydroquinonic and possibly carboxylic groups and also in direct combination with carbon atoms (Puri, 1970). Nitrogen is present in most carbons and coals but the quantities are not considered to be significant. The sulphur present in carbons and coals is fixed as C-S-C or C-S-S-C groups.

Friedel and Pelipetz (1953), Cannon (1953), Van Vucht *et al*. (1955), Brown (1955), Friedel and Queiser (1956), Burvas *et al*. (1970), Schafer (1970), and Shih *et al*. (1972) published coal structure studies by Infrared spectra. Most organic and inorganic materials show absorptions in Infrared region and, in all but a few cases, this absorption includes several characteristic wavelengths. Empirical correlations of vibrating groups with specific observed absorption bands offer the possibility of chemical identification and, coupled with intensity measurements, of quantitative determinations (Willard *et al*., 1965). Brown (1955) studied spectra from 650 to 4000 cm^{-1} for twelve coals obtained from sources which were geographically widely separated and observed a decrease in the number of hydrogen bonded OH groups with increase in carbon content, accompanied by an increase in aromatic hydrogen and a consequent reduction in the aliphatic material. He assigned the absorption frequencies as follows:

Frequency (cm^{-1})	Assignment
3500 (w)	Hydrogen bonded OH/NH
3300 (s)	Hydrogen bonded OH/NH
3030 (w)	Aromatic C-H stretching
2920, 2850 (s)	Aliphatic CH_2 and CH_3 stretching
1605 (s)	Aromatic ring frequency
1160-1300 (s)	C-O stretching of phenols, aromatic ethers, OH
1060-1160 (m)	C-O stretching of alcohols, linear and cyclic ethers
1030 (m)	C-O stretching of aromatic ethers of the type $Ph.O.CH_2R$
860, 820, 750 (w)	Aromatic C-H out of plane vibration frequencies observed in a number of single and condensed ring structures

(s = strong, m = medium, w = weak)

2.6.2 Surface Groups on Silica

Only two kinds of 'end groups' are possible on the surface of silica, silanol groups and siloxane groups

→Si-OH	⇸Si-O-Si⇷
Silanol group	Siloxane group

Siloxane group is chemically less active in comparison with silanol group. Carmon (1940) found that silica particles

acquire a negative charge in alkaline media. This was attributed to the ionization of the silanol groups

$$\rightarrow Si-OH \rightleftharpoons \rightarrow Si-O^{-} + H^{+}$$

In very acidic solution, the surface adsorbs protons and becomes positively charged

$$\rightarrow Si - OH + H^{+} \rightleftharpoons \rightarrow Si - OH_2^{+}$$

Simon and McMahon (1953) from their investigations established the valence stretching

$$\overleftarrow{Si} - \overrightarrow{O} - \overrightarrow{Si} \quad \text{vibration at } 1095\ cm^{-1}$$

2.7 Dual- and Multi-Media Filters

The basic filtration equation of Iwasaki, $-\partial C/\partial L = \lambda C$, indicates, as already stated, that the quantity of removal per unit depth of the filter is proportional to the local concentration of the particles. This results in a logarithmic decline of concentration C with distance L from the inlet face of the filter. Therefore, even if the layers at increasing values of L are removing the same fraction of the concentration reaching them, the removals will be of smaller magnitude. If it is aimed that all the layers of the filter remove greater quantities of particles, then the layers at increasing L, must be designed to remove greater fractions. From the equation of combined transport mechanisms

(Equation 24), it can be seen that one way to do this is to decrease the grain size in the direction of flow. On the contrary, however, in a conventional single-media rapid sand filter, the sand particles are hydraulically graded during backwashing, with the result that the finest particles are at the top of the filter. Because of this, most of the coarse suspended particles are removed at or near the surface of the bed due to predominant contribution by mechanical straining, gravity settling and interception, all mechanisms very much dependent on particle size. A good number of particles passing through the top few inches of the bed will pass through the filter as the above mentioned mechanisms become less predominant in lower levels since the particles are comparatively smaller in size. Therefore, the particles may not be removed to the same extent as in the top layers. An ideal filter would have this stratification reversed so that the pore size would be the largest in the top layer and steadily decrease in size to the bottom layer of the filter; thus the burden of removal would occur in the coarse upper media and the polishing in the fine lower media.

The use of dual-media filters using coarser anthracite coal over finer silica sand was the first major step towards the ideal filter, and the use of such filters is widespread (Cleasby and Woods, 1974). The more recent development of mixed-media or multi-media filters with anthracite coal, silica sand, and garnet sand approach closer

to the ideal pore size arrangement. The densities of these three media vary and with proper selection of particle sizes, the filter will stratify with the anthracite coal in the top, silica sand below the coal, and garnet sand (if used) on the bottom. Due to a lack of suitable anthracite deposits, use of bituminous coals in dual-media filter has been recommended in India (Paramasivam et al., 1973).

Dual or multi-media filters results in larger filter runs and, in some cases, better effluent quality than single-media sand filters (Baumann and Oulman, 1969, and Cleasby and Woods, 1974). Improved effluent quality is apparently due to better attachment potential of the anthracite coal layer inspite of supposedly more difficult transport conditions prevailing in that layer compared to a sand filter due to larger media size.

3. SCOPE OF THE INVESTIGATION

From a review of the recent literature, it is evident that studies on virus removal by filtration were very few. Further, these studies were mainly concerned with the gross removal efficiencies without much attention given to the mechanism involved in such removals. Obviously, there is a great need to obtain more information on the basic mechanisms involved in virus removal by filtration so as to enable the engineer to put the process on a sound scientific basis. With the recent advancement in the concept and theory of water filtration, it is now possible to analyse the experimental results more rationally. The concepts of transport and attachment in filtration, in addition to the realization that the virus particles may exist either as discrete particles or remain attached to the other larger particles constituting turbidity, facilitate in attempts to delineate the removal mechanisms. The removal of viruses from water supplies is extremely important and, becomes more so, with the potential need for wastewater reuse. Information on the basic mechanisms involved in virus removal by filtration should contribute significantly to the solution of the problem and aid in the development of sound design standard for the removal of viruses and other submicron particles in water treatment and wastewater renovation facilities.

Of the important transport mechanisms, diffusion most probably would be the predominant mechanism for the discrete virus particles or similar submicron particles and the total filter depth contributes in such removals. For the virus particles attached to the larger particles, the important mechanisms could be gravity settling, mechanical straining, and interception. By these mechanisms, the larger particles are removed in the top layers and the smaller particles pass through onto the bottom layers of the filter. However, these smaller particles may not be removed to the same extent as the bigger particles. Of the attachment mechanisms, the double layer interactions lose predominance in water filtration. Van der Waal's forces and double layer interaction operate over the same range (about 50 nm or less). If a cationic polyelectrolyte coated and a plain bed are used for particle removal under identical transport conditions, the coated bed is likely to give far better removals as the positive potential of a coated bed adds to the Van der Waal's forces making the bed highly capable in adsorbing the particles compared to the plain filter bed. In addition, mutual adsorption due to polyelectrolytes, polyvalent cations, and polyhydroxy complexes of coagulants, such as alum, may greatly improve the filter performance. In addition to the transport mechanisms governed by the physicohydraulic parameters and the attachment mechanisms governed by the physicochemical parameters of the system, Agrawal (1966) proposed the electrokinetic mechanism which

by itself may be a complete removal mechanism.

The present research was initiated to investigate relative predominance of different mechanisms as stated above in virus removal by filtration. The study was undertaken along the following lines:

a. delineation of the dominant transport mechanisms for the removal of discrete virus particles or similar submicron particles in filtration,

b. estimation of the range of removal of discrete virus particles under various conditions of filter media, suspension, and filtration rates, using coal and sand as filter media,

c. studying the removal of viruses in association with larger particles for delineating the major transport mechanisms in such removal and to estimate the range of removal under various conditions of suspension, and filter media, and

d. comparing the virus removal potentials of single-media rapid sand filter with dual-media coal-sand filter.

4. MATERIALS AND METHODS

4.1 Materials

4.1.1 Virus

Bacterial virus MS2 (MS2 phage) against <u>Escherichia coli</u> was selected as the model virus for this study because of its resemblance to human enteroviruses (polio, coxsackie, and echoviruses) in size, shape, and type of nucleic acid. Enteroviruses belong to the <u>picornavirinae</u> subfamily of the <u>napoviridae</u> family of the naked ribonucleic acid (RNA)-containing viruses with cubic symmetry whereas MS2 phage belongs to the <u>androphagovirinae</u> subfamily of the same family (Goodheart, 1969). MS2 phage has a single stranded RNA core surrounded by a lipid-free protein coat (isoelectric point, 3.9). It is polyhedron in structure having a diameter of 25 nm, and molecular weight of 3.7×10^6 (Overby <u>et</u> <u>al</u>., 1966). MS2 phage is similar to poliovirus in coat protein composition except that the amino acid residue histidine is absent (Table 9).

Precise information regarding the effect of pH on the electrophoretic mobility of MS2 phage (Overby <u>et</u> <u>al</u>., 1966) also allows for a more detailed examination of its behavior in filtration in terms of the mechanisms of removal. Further, it was believed that the difference in properties and behavior among the naturally occurring viruses of concern, i.e. enteroviruses and those between the wild and attenuated

TABLE 9

AMINO ACID COMPOSITION OF POLIOVIRUS AND MS2 PHAGE COAT PROTEINS

Amino Acid	Poliovirus Type 1[φ] (average) (Levintow and Darnell, 1960)	MS2 Phage[φφ] (Fraenkel-Conrat, 1968)
Aspartic Acid	11.9	14 (including aspargine)
Threonine	9.1[⊕]	9
Serine	7.0[⊕]	13
Glutamic Acid	7.7	11 (including glutamine)
Proline	7.2	6
Glycine	6.8	9
Alanine	7.8	14
Valine	7.2	14
Methionine	1.5	2
Isoleucine	4.8	7
Leucine	8.5	8
Tyrosine	3.9	4
Phenylalanine	4.4	4
Cystine (½)	0.8	2
Tryptophan	Destroyed during hydrolysis	2
Lysine	4.7	6
Histidine	2.4	-
Arginine	4.7	4

φ Moles per 100 moles of amino acid recovered
φφ Number out of a total of 129 amino acids
⊕ Corrected for partial decomposition during hydrolysis.

laboratory strains of such viruses would offset the differences in behavior between those viruses and MS2 phage. Consequently, data on the removal of MS2 phage by filtration would be of practical significance. It is interesting to note that a study undertaken by the Metropolitan Water Board, London (1971-73) on virus removal by slow sand filtration showed practically identical results with poliovirus and MS2 phage. It was also observed that 'these results indicate that MS2 is a very good virus indicator for use in filtration studies......'

4.1.2 Biological Media

L Broth

(Constituents per liter of water)

Bacto-Tryptone (Difco)	10.0 gm
Yeast Extract (Difco)	5.0 gm
NaCl	10.0 gm
Glucose	1.0 gm
2 <u>M</u> $CaCl_2$	1.0 ml

Adjusted to pH 7.0 with 1 <u>N</u> NaOH

L Agar

L Broth plus 15 to 20 gm/l Agar Agar (Japan)

Soft Agar

L Broth plus 10 gm/l Bacto Agar (Difco)

Phage Buffer

(Constituents per liter of water)

Tris(hydroxy methyl)aminomethane	6.06 gm
NaCl	5.85 gm

Adjusted to pH 7.6 with 1 N HCl

4.1.3 Glassware

All glassware were soaked overnight in 0.3 percent B-300 Teepol (Manufactured by Surfactants Private Limited, Bombay, India) followed by rinsing in tap water and distilled water, as suggested by Rao (1971). Sterilisation was accomplished in a hot air oven at 180°C for 2 hr or longer.

4.1.4 Filter Media

4.1.4.1 Sand

The sand used was silica sand obtained from the Kanpur Water Works. It was washed several times with tap water, dried at 103°C for 24 hr, seived to a geometric mean size (G_m) of 0.5 mm and stored for future use. Sphericity of this sand was 0.8 and porosity in the filter column was 0.45.

4.1.4.2 Coal

The coal used was high-grade Giridih bituminous coal obtained through the National Environmental Engineering Research Institute, Nagpur. This was selected because of its excellent virus sorptive capacity among

several Indian bituminous coals tested (Oza, 1974). Also Giridih bituminous coal possesses superior physical properties for use as a filter media among other Indian bituminous coals examined (Table 10).

The coal was crushed and seived to geometric mean sizes (G_m) of 0.5 and 1.0 mm, washed several times in tap water, dried at 103°C for 24 hr and stored for future use. Sphericity, and porosity in filter column were 0.74 to 0.76 and 0.5, respectively.

4.1.5 Polyelectrolytes and Chemicals

High molecular weight, water soluble, synthetic organic polyelectrolytes, Purifloc A-23 (anionic) and purifloc C-41 (cationic) manufactured by Dow Chemical Co., Middland, Michigan were used. Other chemicals used were alum ($Al_2(SO_4)_3 \cdot 16H_2O$) and sodium chloride marketed by Sarabhai M. Chemicals Ltd., Baroda, and calcium chloride ($CaCl_2 \cdot 2H_2O$) marketed by Riedel-De Haen AG Seelze, Hannover, Germany. All these chemicals were of analytical reagent grade.

4.1.6 Water

Except for the batch sorption tests, water from Tube Well No. 4, Indian Institute of Technology, Kanpur was used. Typical chemical analysis of the water is shown in Table 11. Glass distilled water was used in the batch sorption tests.

TABLE 10

PHYSICAL PROPERTIES OF INDIGENOUS HIGH GRADE BITUMINOUS COALS

Characteristics	Range of Values for Indian Bituminous Coals (Technical Digest, 1971)	Values for Giridih Coal (Oza, 1974)	Values for Churcha Coal (Oza, 1974)	Remarks
1. Specific Gravity	1.3-1.5	1.33/1.465 φ	1.35	AWWA specifies 1.55 to 1.65 for anthracite
2. Moh's hardness	Around 3	2-3	2-3	AWWA specifies 3 to 3.75 for anthracite
3. Percent Loss in Weight After 100 hr Backwashing	Less than 5	0.63	0.80	Comparable to anthracite
4. Solubility in Dilute HCl	Generally less than 2	0.69	1.87	Anthracite is generall inert
5. Sphericity	0.63-0.79	0.74-0.76	0.76-0.77	Recommended minimum value for filter media is 0.6
6. Phenol Leaching	Not detectable	Not detectable	Not detectable	Permissible limit in drinking water is 0.001 mg/l

φ Author's value for the coal sample used.

TABLE 11

CHEMICAL ANALYSIS OF TUBE WELL WATER

Constituents	Concentration, in milligrams per liter (except pH)	
pH	7.8 to 8.2	
Conductance	770 to 980 µmhos/cm	
Alkalinity (HCO_3^-)	440 to 450	(as $CaCO_3$)
Hardness	200	(as $CaCO_3$)
Calcium	46 to 50	(as $CaCO_3$)
Magnesium	150 to 154	(as $CaCO_3$)
Chlorides	20 to 21	
Sulphates	24 to 25	
Phosphates	0.1 to 0.2	
Iron	0.2 to 0.3	
Manganese	0.2 to 0.3	
Silica	23 to 24	
Sodium + Potassium	180 to 185	
Turbidity	Not detectable	

4.2 Equipment and Accessories

A laboratory bench-scale filtration apparatus was employed in this study. It consisted of a glass filter column (pyrex glass), 2.54 cm (1 in.) ID and 1 m long, with suitable ports for virus and turbidity sampling, headloss measurements, and potential probes for streaming potential measurements, at 15.23 cm (6 in.), 38.1 cm (15 in.), 45.7 cm (18 in.), 61 cm (24 in.), and 83.8 cm (33 in.) from the top. Any port not in use during an experiment was kept closed with a polyethylene or glass plug. A 60 l plastic container equipped with a stirrer served as the overhead storage tank and provided a filtration head of 250 cm. The filtration rate was monitored through a flow meter and kept constant during an experiment by manually adjusting the filter outlet. Glass manometers, 0.4 cm OD, were used for headloss measurements.

The filter column was used for the measurement of streaming potentials of filter media. It also facilitated studies on the effect of backwashing on the streaming potential of coated filter media. Platinum electrodes (0.03 cm diameter), 5 cm long, fixed into male ground glass plugs and fitted to the filter column through female ground glass joints were used. The ground glass joints were made watertight. Platinum is quite resistant to poisoning or polarization in natural waters, and dilute suspensions and solutions which are not strongly reducing or oxidizing (Agrawal, 1966). The electrodes were properly connected to a millivolt meter (Beckman

Expandomatic pH meter, Model 76005) having 1 mV divisions on the expanded scale. In this scale the range could be selected over any 200 mV span between -2000 mV to +2000 mV and potentials could be read upto 0.25 mV by interpolation. The millivolt meter was provided with manual temperature compensator and high impedence vacuum tube system necessary to obtain accurate measurement of small potential across a high impedence filter system.

A conductivity meter (Wiss-Techn Werkstatten, Weilheim obb, Germany) having a range of 10^{-2} to 10^{-8} mhos was used for conductivity measurements. The cell constant was determined using standard KCl solution (Standard Methods, 1971). Particle electrophoretic mobility values were measured using a Zeta meter (Zeta Meter Inc., New York). A Philips pH meter (Model PR9405M) was used for pH measurements. For turbidity measurements, a Hellige turbidimeter (Hellige, Inc., New York) calibrated in Formazin turbidity units (FTU) (Standard Methods, 1971) was employed.

4.3 Methods

4.3.1 Preparation and Assay Procedure for MS2 Phage

The initial stock cultures of MS2 phage and its host, *Escherichia coli* A19, were obtained from the Environmental Engineering Laboratory, University of Illinois. Subsequent suspensions were prepared according to the following procedure.

An 1-liter flask containing 900 ml of L Broth and equipped with an aeration device was inoculated with overnight grown Escherichia coli A19 so as to get an A_{660} of 0.06 to 0.10. The flask was maintained at 37°C and aerated until the cell culture reached an A_{660} of 0.15 to 0.20. This corresponds to an early log growth phase of Escherichia coli A19 (5×10^{7}/ml). The required amount of MS2 phage stock was then added at a multiplicity of 6 to 10. The aeration of the flask at 37°C was continued with A_{660} measurements at intervals. First the absorbance would increase and later it might level off or gradually decrease indicating lysis. Five ml of chloroform was added after lysis had started and the lysate was stirred at a high speed for 1 min using a magnetic stirrer. It was then kept at 4°C and stirred every 5 min. After 30 min, the lysate was centrifuged at low speed (5,900Xg for 10 min) to remove bacterial cell debris followed by high speed centrifugation (23000Xg for 2½ hr) in a super speed refrigerated centrifuge (Janetzki Model K24). The virus pellet was resuspended in 5 ml of phage buffer and titered. The suspension was then stored at 4°C for future use

Soft agar technique of Adams(1959) as followed by Chaudhuri (1969) was adopted for enumeration of MS2 phage.

Before assaying, the sample was diluted in L Broth to yield 100 to 300 plaques per plate. A liquid top-agar mixture consisting of about 3 ml of soft agar at 45°C, 0.3 ml of a log growth phase culture of Escherichia coli A19 cells,

and 0.1 ml (unless otherwise specified) of the diluted virus sample was plated on solidified bottom agar (L Agar) plates and incubated at 37°C for 6 to 8 hr. Plaques were counted with the aid of a Quebeck colony counter and reported as plaque forming units per ml (PFU/ml). Triplicate plates were prepared from each sample to increase accuracy.

4.3.2 Agitated Non-Flow Experiments

To study the potential of sand and coal as sorbents in the removal of viruses, batch sorption tests were carried out in non-flow agitated systems using a rotating shaker. Experiments were conducted at pH values 6.0 (0.2 M maleate buffer), 6.8 (0.2 M maleate buffer), and 8.4 (0.2 M borate buffer). The reaction mixture (total volume 900 ml) was prepared by adding 45 ml of buffer solution and 835 ml of glass distilled water. The ionic strength was adjusted to 0.02 by adding 1 N NaCl. MS2 phage stock was suitably diluted and added to the reaction mixture to give the required input virus concentration. The reaction mixture was then thoroughly stirred and sample taken for initial virus concentration. Then 295 ml aliquots were transferred to each of the three 300 ml bottles. One bottle served as the control (no sorbent) and the other two bottles served as reactors with either 10 g/l of coal or sand as sorbent. All experiments were performed at room temperature (25°C to 27°C).

4.3.3 Streaming Potential of Filter Media

As the streaming potentials generated are directly proportional to headloss across the filter depth under consideration, the potential differences and corresponding headlosses were measured. Also, the conductivity of the water was determined for calculating zeta potentials. Significant zero potentials were observed as reported by Gregory (1964), Oulman and Baumann (1964 and 1964a) and Agrawal (1966). Potential reading was taken at a particular flow rate and headloss and then the flow was shut off and the potential reading was taken again allowing as little time to lapse as possible. The difference gave the correct streaming potential as the headloss and consequently the streaming potential would be zero under no flow condition. As recommended by Gregory (1964) and Agrawal (1966), the potential was recorded after maintaining the flow rate for 15 min or more to establish a steady state. In this work, surface conductance corrections were not applied as they were insignificant because of the high conductivity of the water. Also, due to the high conductivity of the water used in this investigation, at each flow rate only one set of headloss and potential measurements was possible. The streaming potential and headloss measurements were taken for a number of flow rates and the average P/H value was obtained for zeta-potential calculation.

4.3.4 Coating of Filter Media with Polyelectrolyte and Alum

To alter the zeta-potential of filter media, the media was treated _in situ_ with polyelectrolyte. For polyelectrolyte coating, purifloc C-41 or A-23 was put into the required volume of water and the suspension was run through the filter bed at 4.9 m/hr (2 gpm/sq ft) for about 15 min. When the suspension was about 1 in. above the top of the bed, the flow was stopped and the media allowed to stand in the polyelectrolyte suspension for about 30 min. Water was then admitted through the filter outlet and the bed was suspended without turbulence thus giving a mild backwash for about 1 min. The bed was then allowed to settle and subsequently flushed with water at 4.9 to 9.8 m/hr (2 to 4 gpm/sq ft) till the headloss recorded was equal to that before coating. For alum floc coating of filter media, the following procedure was adopted. The required quantity of filter media and alum solution (50 mg/l) were taken into a 300 ml plastic bottle and kept on a rotating shaker for 1 hr. At this high dose, alum formed micro flocs and coated the media. The media was then carefully transferred into the filter column without agitation. This procedure ensured a uniform coating of filter media.

4.3.5 Preparation of Stock Turbid Suspension

In order to obtain clay suspensions of uniform particle size, 3 g of kaolinite in 1 litre of water was vigorously stirred for about 30 min and allowed to settle for 2 hr. The middle third of the supernatant was then siphoned out which served as the stock turbid suspension from which turbid influent waters for filtration experiments were prepared

4.3.6 Filtration Experiments

A set procedure was followed for the filtration experiments. Prior to initiating an experiment (filter run), a 0.3 percent B-300 Teepol solution was introduced into the filtration system and retained in the apparatus for about 2 hr. Then the solution was removed and the system washed with water. Teepol was used to coat the system so as to minimize the sorption of viruses on the exposed surfaces. The filter column was then filled with water and the required amount of filter media (coal or sand) for a desired depth of filter bed placed in the column. For filter runs with polyelectrolyte coated filter media, the bed was coated _in situ_ with polyelectrolyte. Influent water containing the desired concentration of the model virus was then placed in the overhead storage tank and the experiment initiated at a desired filtration rate by adjusting the filter outlet while monitoring the flow through a flowmeter. To avoid any disturbance due to sudden change

in flow, the sampling ports under use during a filter run were adjusted to flow at a constant rate of about 1 ml/min. Head loss measurements were taken at various depths and time intervals during the run. Samples for virus enumeration and turbidity were taken at desired depths and at the filter outlet while influent samples were collected form a port located immediately above the filter bed. Samples for virus enumeration were immediately diluted in L broth, stored at 4°C, and plated as soon as possible but always within 2 hr of collection. Influent and effluent samples were also occasionally checked for pH to record any change in pH during a run. Filter runs were usually of 8 hr duration or less except for those designed to simulate practical filtration conditions which were continued upto 24 hr or longer.

Filter runs were performed with the model virus suspended in turbidity-free as well as turbid waters, and alum coagulated and settled water containing the model virus. For filter runs with turbid waters, the influent water in the storage tank was stirred occasionally (about every 30 min). This was found adequate in minimizing settling or appreciable change in the size of the clay particles during a 5.5 hr filter run when initial turbidity was below 25 FTU. From background experiments it was observed that a synthetically prepared turbid water (70 to 90 FTU) could be clarified to a final turbidity of 30 to 45 FTU when coagulated with 7 to 8 mg/l of alum followed by 30 min settling. For filter runs with alum

coagulated and settled water, 60 litres of turbid water (70 to 90 FTU) containing the desired initial concentration of the model virus was coagulated with 7 to 8 mg/l of alum, using 1 min flash mix, 20 min slow mix at 20 rpm followed by 30 min of settling. The supernatant was then transferred to the storage tank. Samples for turbidity and particle size measurements, and virus enumeration were taken before and after coagulation and settling. Considerable variation in influent turbidity and virus concentration was observed as the filter run progressed in spite of continuous stirring of the influent water in the storage tank. Therefore, while calculating filter efficiencies the actual values for influent turbidity and virus concentration at a particular time were used.

Two filter runs were also performed with alum coagulated and settled water containing low initial density (52 - 210 PFU/ml) of the model virus employing a membrane filter concentration technique (Metropolitan Water Board, London, 1971-73). For densities lower than this, i.e. 52 PFU/ml, it was found difficult to enumerate filter effluent virus density. The concentration procedure consisted of adding calcium chloride solution to an 1 liter sample to make it 0.1 <u>M</u>, adjusting its pH to 7.0 with 1 percent hydrochloric acid followed by filtration through a 0.45 μm cellulose nitrate membrane filter (Millipore Corporation, Bedford, Mass.).

The membrane with the adsorbed viruses was then placed in a flask containing 5.6 ml of 3 percent beef extract adjusted to pH 9.0 with 5 N sodium hydroxide. The flask was subsequently placed in a reciprocating shaker for 40 min for virus elution and the eluate plated for virus enumeration. Background experiments revealed that this method produced 99 percent virus recovery.

5. RESULTS AND DISCUSSION - I

The experimental data[φ] on zeta potentials of the filter media (sand and coal) and suspended particles (discrete virus particles and the particulates in turbid water) are presented in this chapter. This information will be used in the subsequent chapters for interpreting the results of the filtration studies.

5.1 Zeta Potentials of Filter Media

The streaming potentials (P) measured are plotted on an arithmetic paper against corresponding headlosses (H) and a line of best fit drawn by visual inspection. The slope of this line gives the average streaming potential in mV/cm of headloss. Zeta potentials at 25°C ($\mu = 0.9 \times 10^{-2}$ poise and $\varepsilon = 78.54$) are computed from the average streaming potentials using the following equation:

$$(\mathrm{mV}) = 1.32\ \Lambda\ \left(\frac{P}{H}\right) \qquad 27$$

where, Λ = conductivity, µmhos/cm

P/H = streaming potential, mV/cm

φ The experimental data obtained in the filtration studies are presented in Appendix-II. Appropriate reference is made in the Fig. and Tables appearing in this and subsequent chapters.

This equation is derived from the original equation of Ranz and Wong (1952):

$$\zeta = \frac{4\pi \ \mu \ \varepsilon_o \ P}{\varepsilon \ R \ H} \qquad 28$$

5.1.1 With Varying Ionic Composition of Water

Streaming potentials of 0.5 mm sand and 1 mm Giridih coal were measured in the tube well water and with varying concentrations of added Na^+, Ca^{++} and Al^{+++} (Fig. 1 to 6). Since the water was highly conductive, at each flow rate only one set of potential and headloss readings were possible (measured over a media depth of approximately 45 cm). It is observed that the streaming potentials of both sand and coal are positive in the tube well water and decrease with addition of increasing concentrations of the cations. The streaming potential of sand did not become negative with Na^+ addition as reported by Gregory (1964) but for coal, it eventually became negative when 0.475 meq/l of Na^+ was added. However, for both sand and coal, decrease in P/H was not adequate to counterbalance the increase in conductance when 0.380 meq/l of Na^+ was added. The positive streaming potentials of both sand and coal decreased with addition of Ca^{++} and became negative for addition of 0.380 meq/l or more.

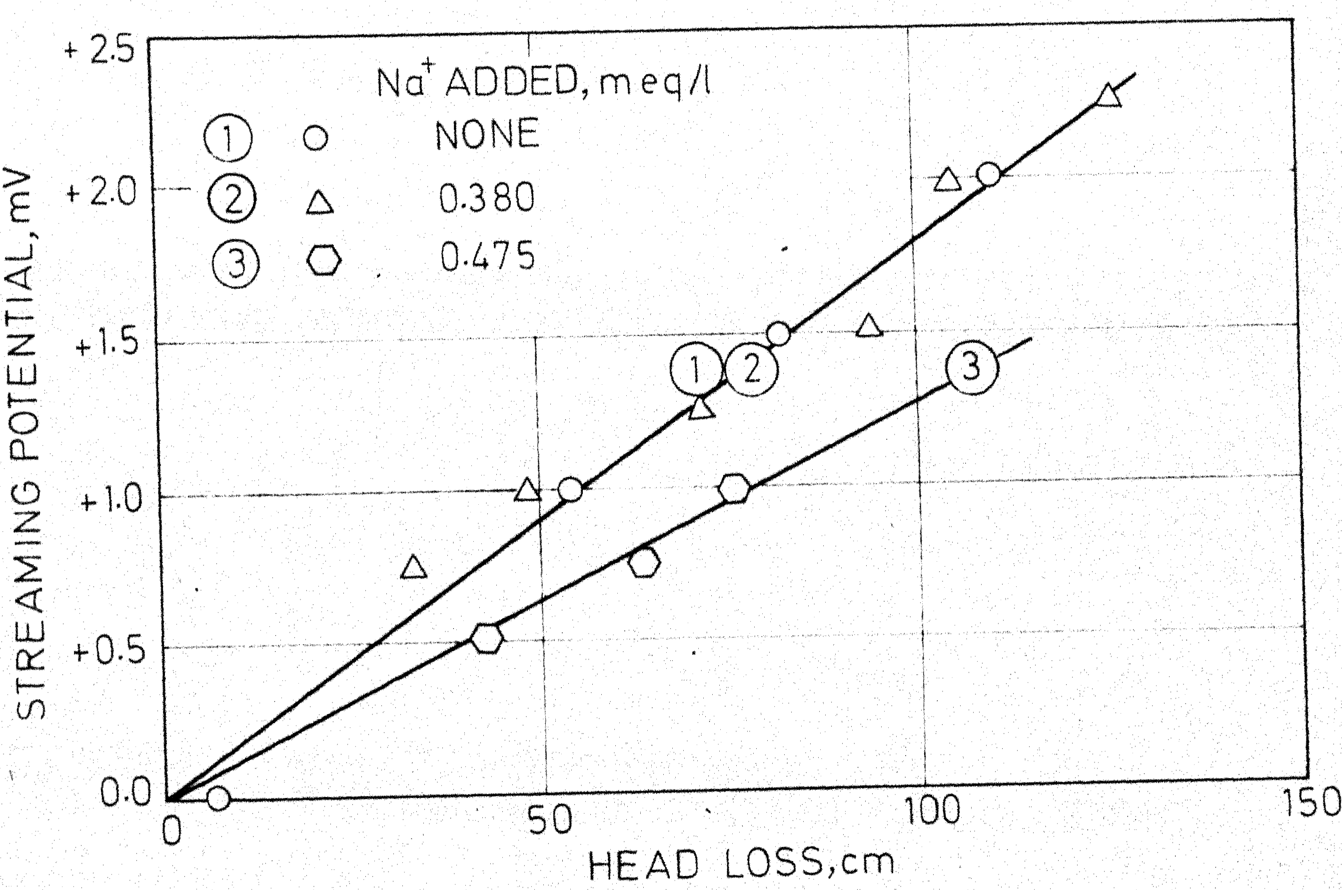

FIG. 1. STREAMING POTENTIALS OF SAND WITH VARYING CONCENTRATIONS OF Na^+ ADDED TO TUBE WELL WATER (Ref: Table A1)

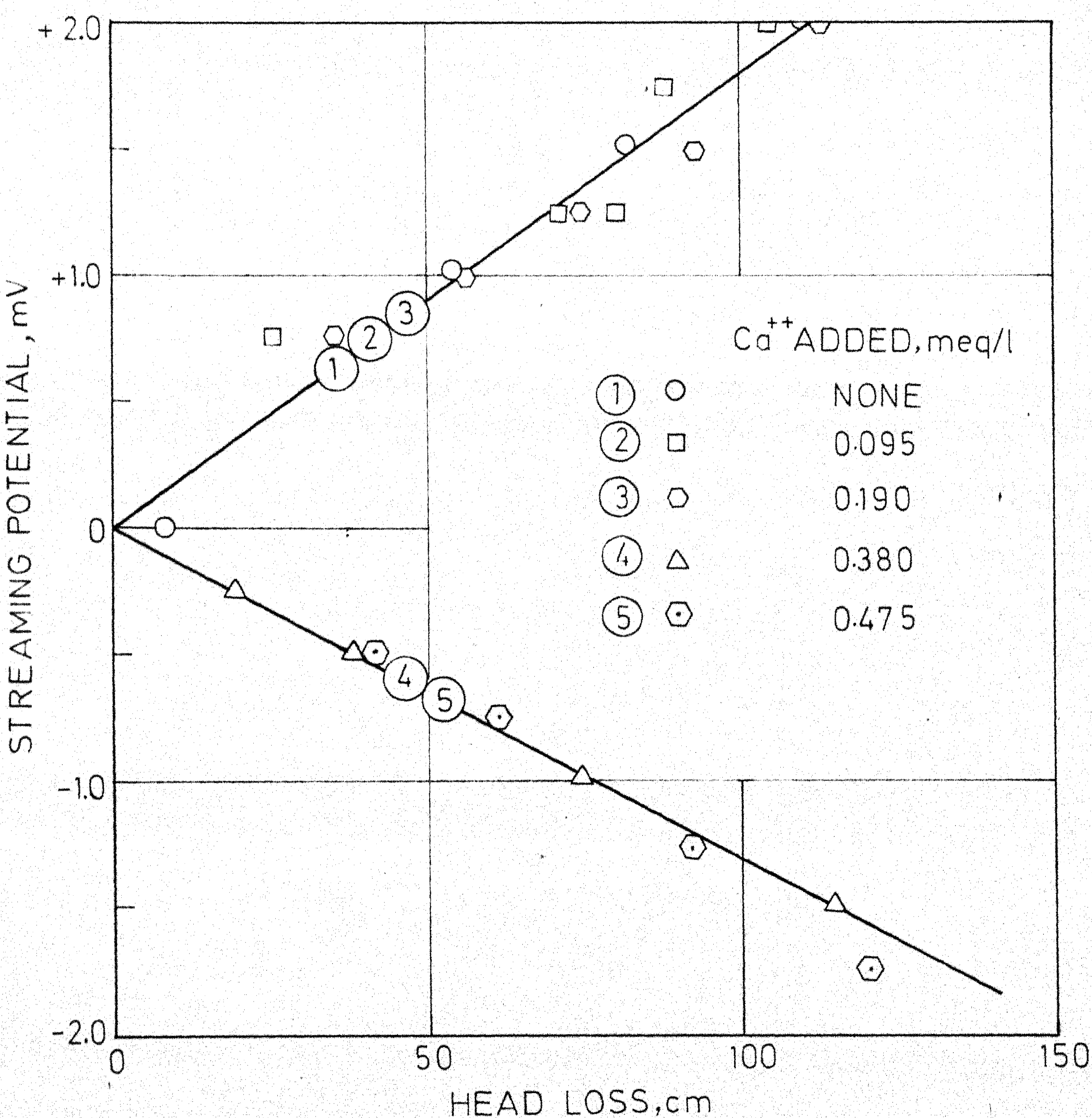

FIG. 2. STREAMING POTENTIALS OF SAND WITH VARYING CONCENTRATIONS OF Ca^{++} ADDED TO TUBE WELL WATER (Ref: Table A2)

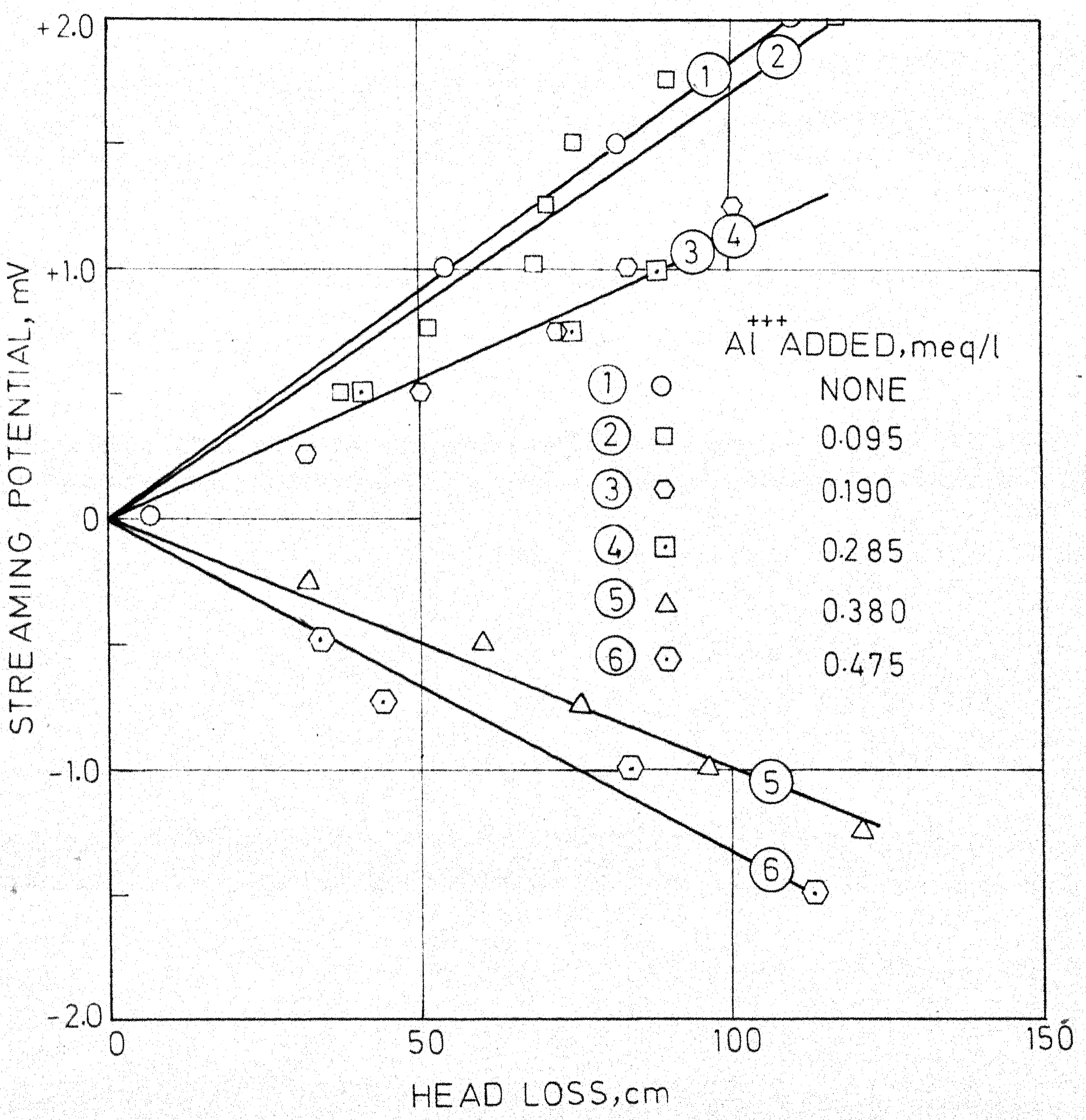

FIG.3. STREAMING POTENTIALS OF SAND WITH VARYING CONCENTRATIONS OF Al^{+++} ADDED TO TUBE WELL WATER (Ref: Table A3)

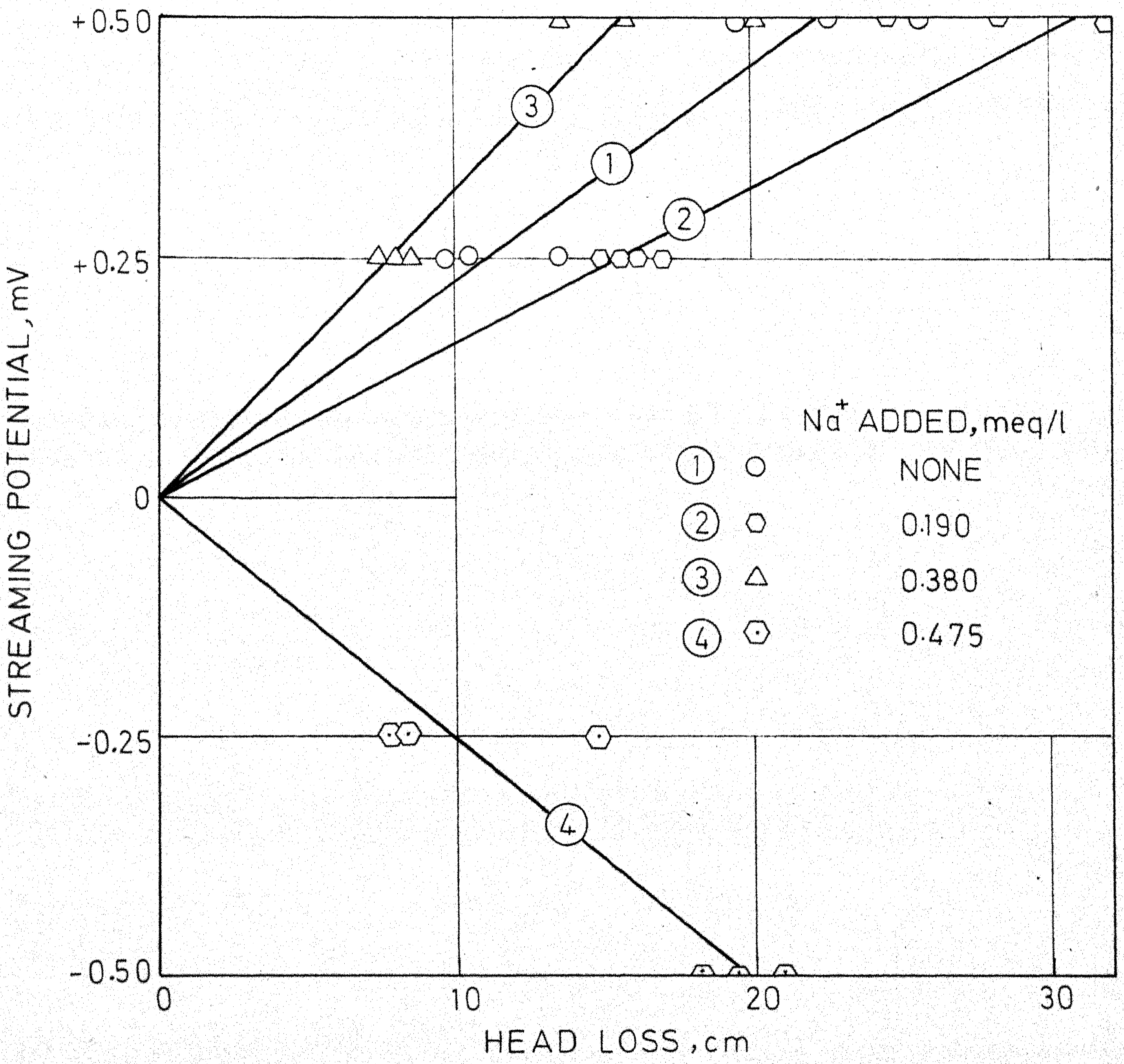

FIG. 4. STREAMING POTENTIALS OF COAL WITH VARYING CONCENTRATIONS OF Na^+ ADDED TO TUBE WELL WATER (Ref: Table A4)

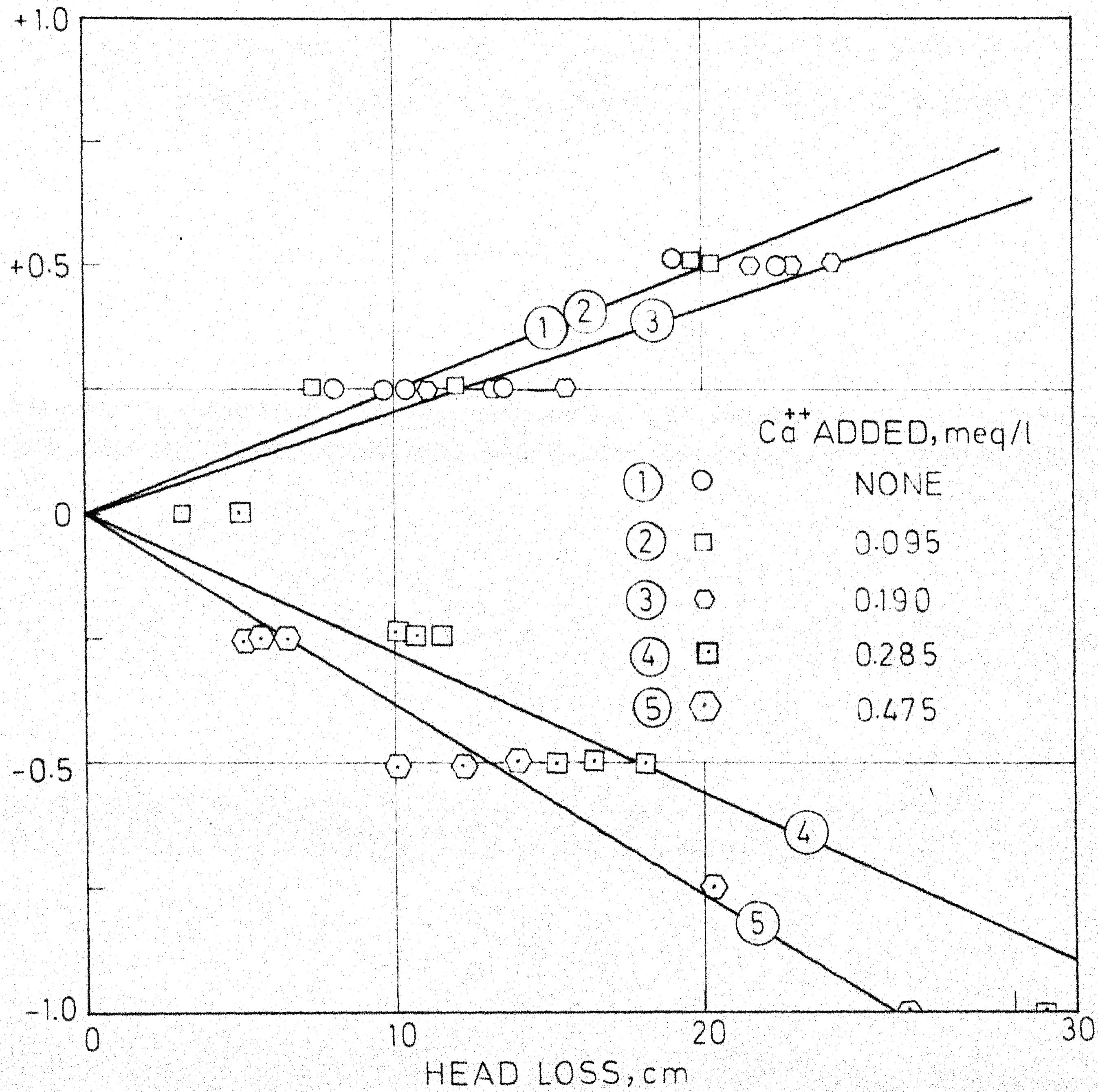

FIG. 5. STREAMING POTENTIALS OF COAL WITH VARYING CONCENTRATION OF Ca^{++} ADDED TO TUBE WELL WATER (Ref: Table A5)

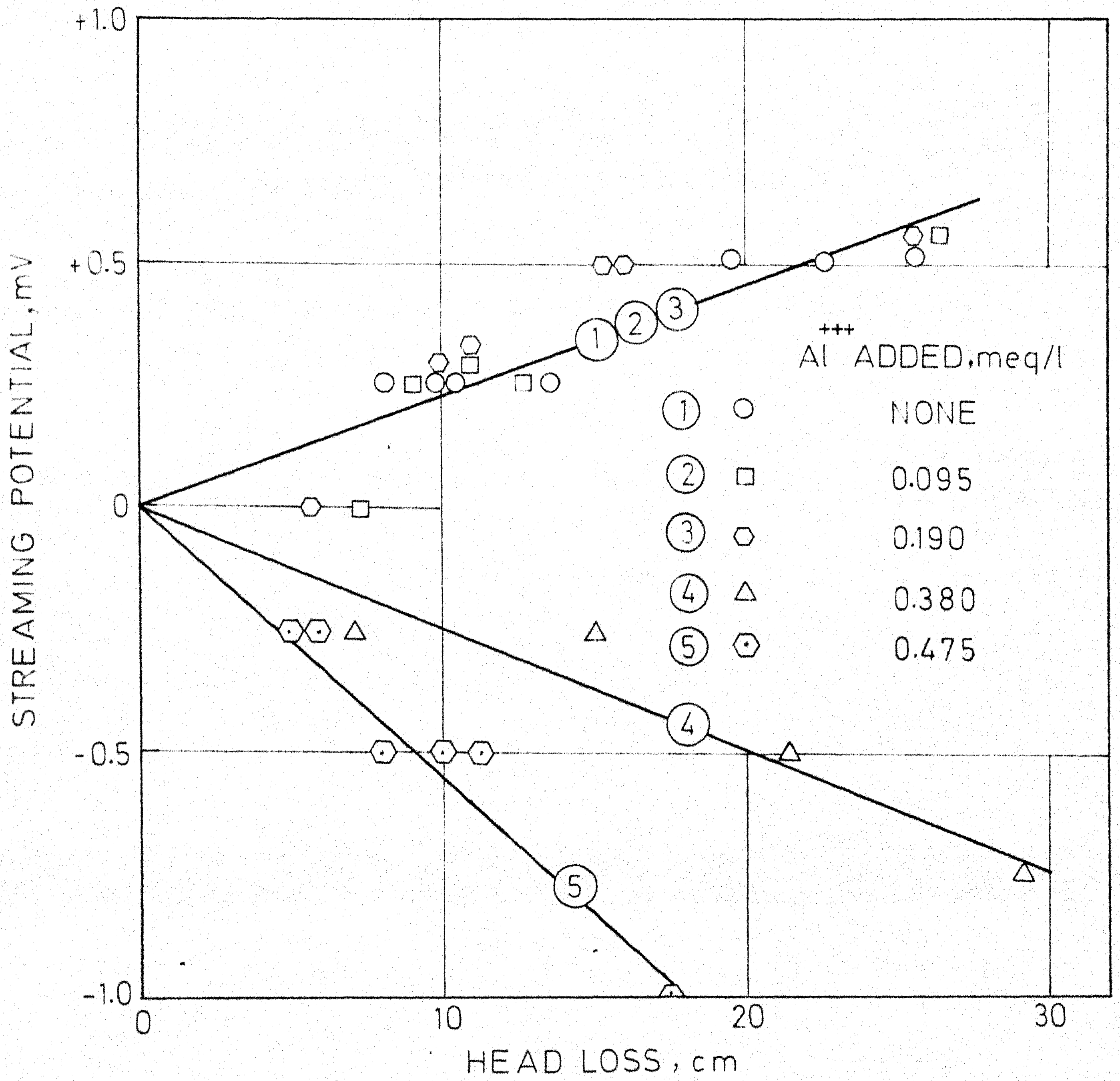

FIG. 6. STREAMING POTENTIALS OF COAL WITH VARYING CONCENTRATIONS OF Al^{+++} ADDED TO TUBE WELL WATER (Ref: Table A6)

Since the streaming potentials are set up as a consequence to shearing off of double layers, they are opposite in sign to the zeta potentials. The computed zeta potentials of sand and coal are shown in Table 12 and Fig. 7. It is seen that eventhough the zeta potential of coal is slightly more negative than that of sand in the tube well water, it is much more positive when cations are added, i.e. coal surface is more responsive to the chemical changes in the surrounding environment. The zeta potential of sand in the tube well water is -18.5 mV which agrees well with the values reported by Gregory (1964) and Agrawal (1966), and is of the same order as observed by Baumann and Oulman (1964, 1970) for diatomaceous earth.

5.1.2 With Alum or Polyelectrolyte Treatment

Alum floc or polyelectrolyte treatment was intended to change the zeta potential of the filter media. The polyelectrolytes used were Purifloc C-41 (cationic) and Purifloc A-23 (anionic). It was expected that when the media was treated with the cationic polyelectrolyte or alum floc, the negative zeta potential would be reduced or possibly reversed in sign, and become more negative when treated with the anionic polyelectrolyte. The streaming potentials measured and the headlosses recorded are shown in Fig. 8 and 9 and the computed zeta potentials are presented in

TABLE 12

ZETA POTENTIALS OF SAND AND COAL WITH VARYING IONIC COMPOSITION OF WATER

Composition of Water	Λ µmhos/cm	Sand		Coal	
		P/H mV/cm	Computed ζ_m mV	P/H mV/cm	Computed ζ_m mV
Tube Well Water	770.0	2/110	-18.50	1/44	-23.10
+0.190 meq/l Na^+	794.0			0.375/23	-17.10
+0.380 meq/l Na^+	818.0	2/110	-19.65	1/40	-27.00
+0.475 meq/l Na^+	830.0	1/80	-13.70	1/40	+27.40
+0.095 meq/l Ca^{++}	782.8	1.25/70	-18.45	1/40	-25.80
+0.190 meq/l Ca^{++}	795.6	1.25/70	-18.75	0.375/18	-21.80
+0.285 meq/l Ca^{++}	808.4			1/36	+29.60
+0.380 meq/l Ca^{++}	821.2	1.25/95	+14.25		
+0.475 meq/l Ca^{++}	834.0	1.25/95	+14.50	1/26	+42.30
+0.095 meq/l Al^{+++}	784.0	1/60	-17.25	1/44	-23.55
+0.190 meq/l Al^{+++}	798.0	1/90	-11.70	1/44	-24.00
+0.285 meq/l Al^{+++}	812.0	1/90	-11.90		
+0.380 meq/l Al^{+++}	826.0	1/100	+10.90	1/40	+27.25
+0.475 meq/l Al^{+++}	840.0	0.875/65	+14.92	1/18	+61.60

(Data Source: Fig. 1 to 6 and Tables A1 to A6).

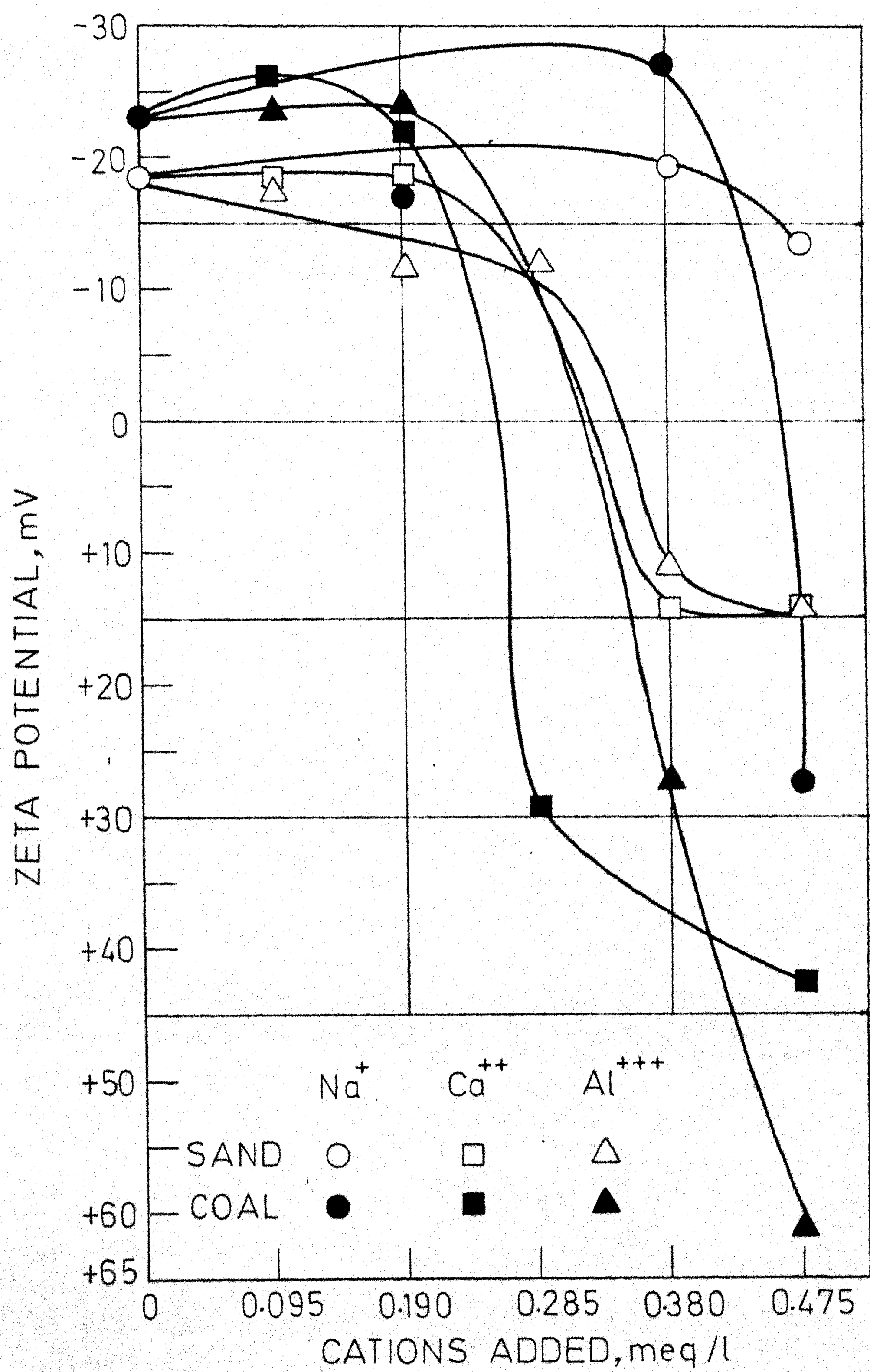

FIG.7 ZETA POTENTIALS OF SAND AND COAL WITH VARYING IONIC COMPOSITION OF WATER (Ref: Table A12)

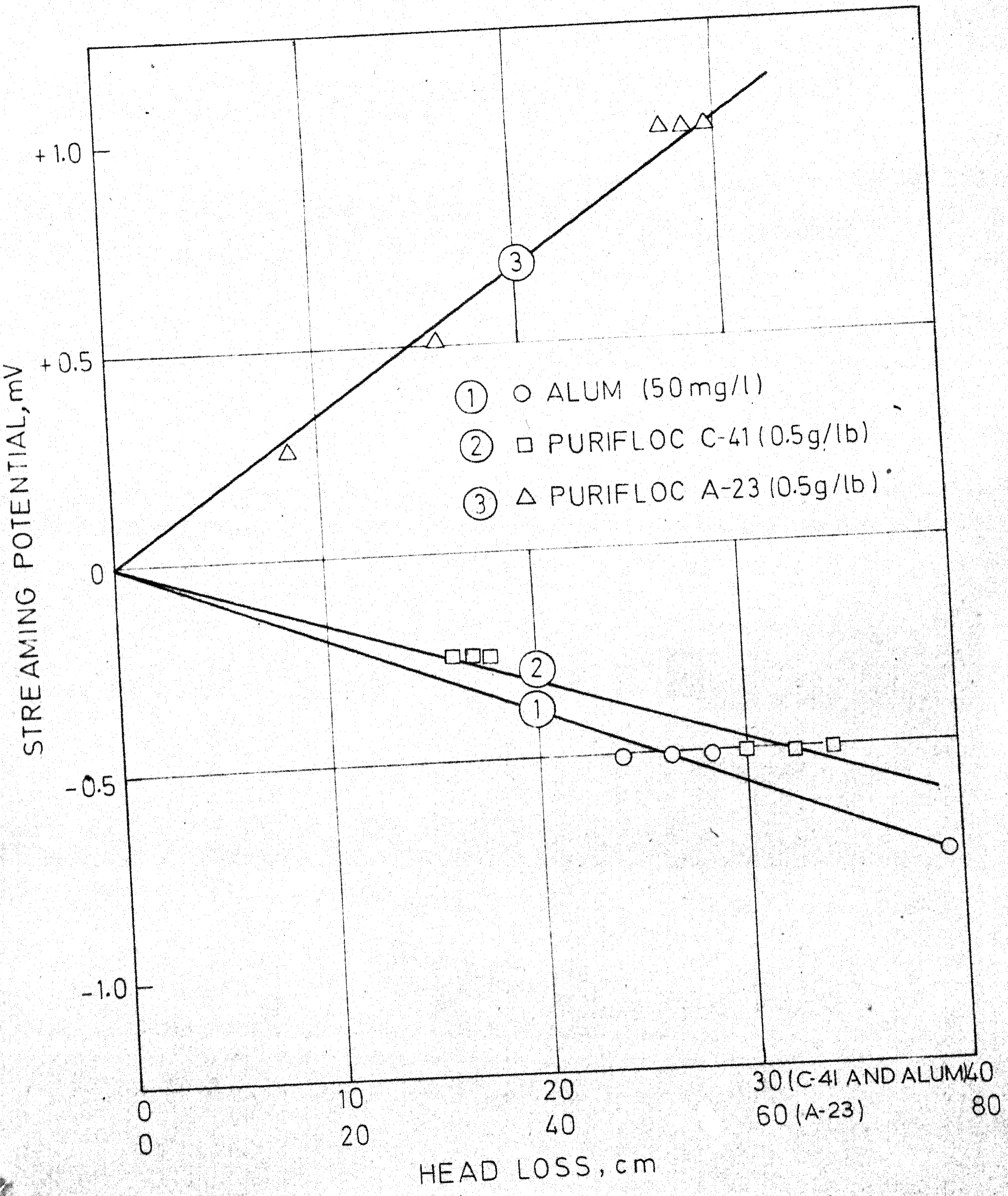

FIG.8 STREAMING POTENTIALS OF SAND AFTER TREATMENT WITH ALUM OR POLYELECTROLYTE (Ref: Table A7)

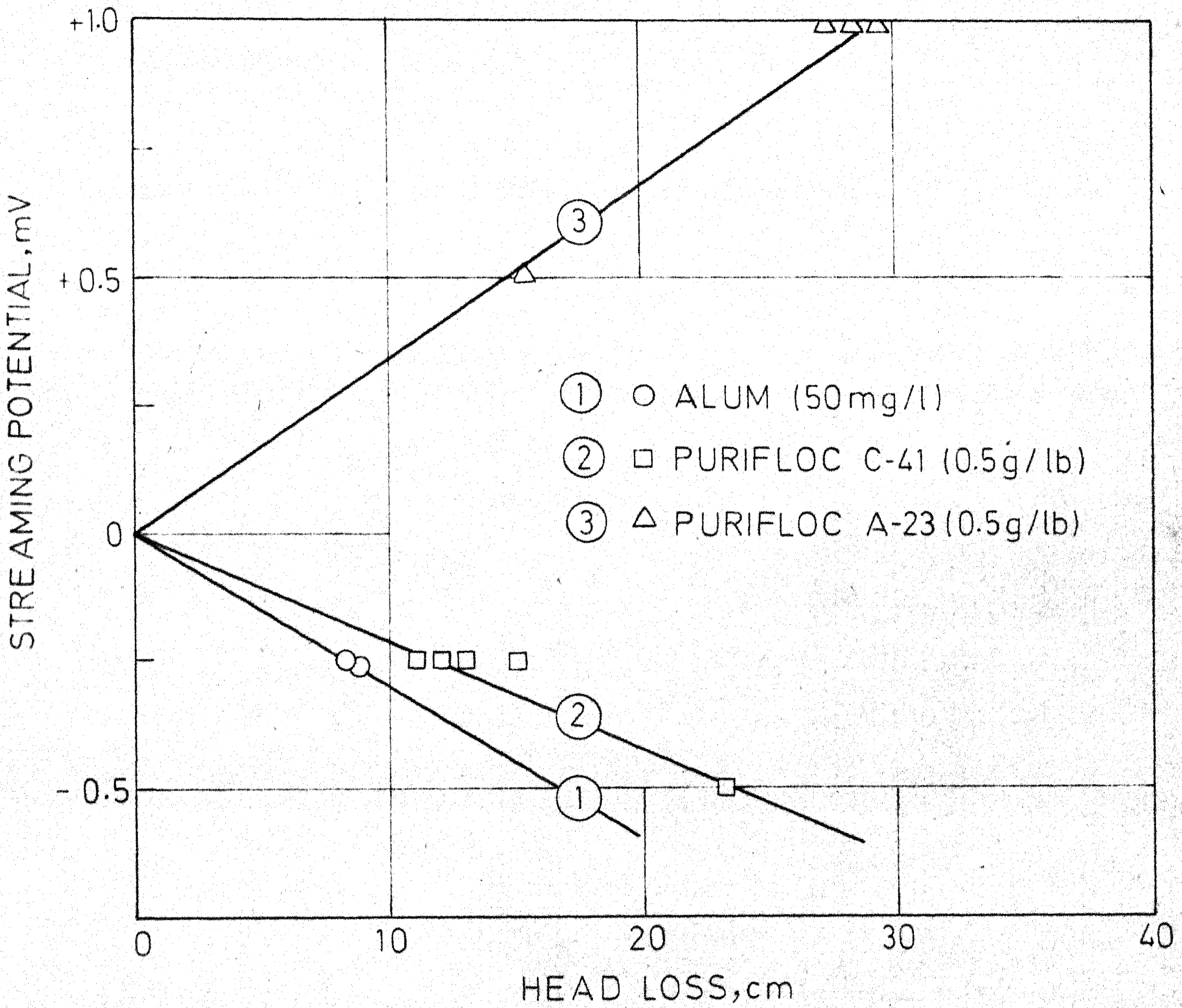

FIG. 9 STREAMING POTENTIALS OF COAL AFTER TREATMENT WITH ALUM OR POLYELECTROLYTE (Ref: Table A8)

Table 13. Both alum floc and Purifloc C-41 reversed the zeta potential of both sand and coal. The zeta potential of Purifloc C-41 or alum treated coal was about 1.5 times of that of sand. However, when coal was treated with the anionic polyelectrolyte, eventhough there was an increase in the negative zeta potential, the increase was much less pronounced compared to that of sand treated similarly. This was probably due to the fact that the coal surface had a high negative zeta potential initially.

Polyelectrolyte dose of 0.5 g/lb of filter media was based on the optimum dosage findings of Baumann and Culman (1970) and Agrawal (1966). However, in this study the objective was to investigate the influence of polyelectrolyte treatment on the surface charge of the media and consequent effect on the removal efficiency of a filter, and not to find out the optimum dosages. It was evident from equal headloss values observed on treated and untreated beds at a particular flow rate that the polyelectrolyte coating was a micro layer in thickness and did not change the flow characteristics in the filter. Also, it was observed that backwashing at 50 percent expansion for 15 min washed away the micro layer reversing the zeta potential to a negative value.

TABLE 13

ZETA POTENTIALS OF SAND AND COAL AFTER TREATMENT WITH ALUM FLOC OR POLYELECTROLYTE

Tube Well Water: Λ = 770 µmhos/cm

Treatment	Sand		Coal	
	P/H mV/cm	Computed ζ_m mV	P/H mV/cm	Computed ζ_m mV
Untreated	2/110	-18.50	1/44	-23.10
0.5 g/lb Purifloc C-41	0.5/32.5	+15.65	0.375/17.5	+21.80
0.5 g/lb Purifloc A-23	1.125/32.5	-35.20	0.25/7.5	-33.80
Alum (50 mg/l)	0.625/32.5	+19.50	0.375/12	+31.80

(Data Source: Fig. 8, 9 and Tables A7, A8).

5.2 Zeta Potentials of Suspended Particles

5.2.1 Virus

The electrophoretic mobilities of discrete virus particles cannot be found with the normal laboratory facilities as the particles can be observed only under an electron microscope. Overby _et al_. (1966) reported electrophoretic mobility of MS2 phage at various pH values (Fig. 10). However, the mobility values given in $cm^2/(sec.\ volt)$ seem to be very high. As the electrophoretic mobility is often expressed in the literature as (micron. cm)/(volt. sec), the reported values were probably $10^{-4} \times cm^2/(volt.\ sec)$. Based on this assumption and using their data, the computed zeta potentials of discrete virus particles at 30°C were -10 mV, -15 mV and -18 mV at pH 6, 7 and 8, respectively. These values seemed to be appropriate by comparing with the values of similar order reported by Jorden (1963) and Agrawal (1966) for particles such as bacteria and therefore, were used in interpreting the results of the present study.

5.2.2 Clay

Agrawal (1966) calculated zeta potentials of suspended particles from the electrophoretic mobilities measured in a modified Brigg's cell. He observed that the mobility and the zeta potential of all the particles tested were negative in distilled as well as in tap water. However,

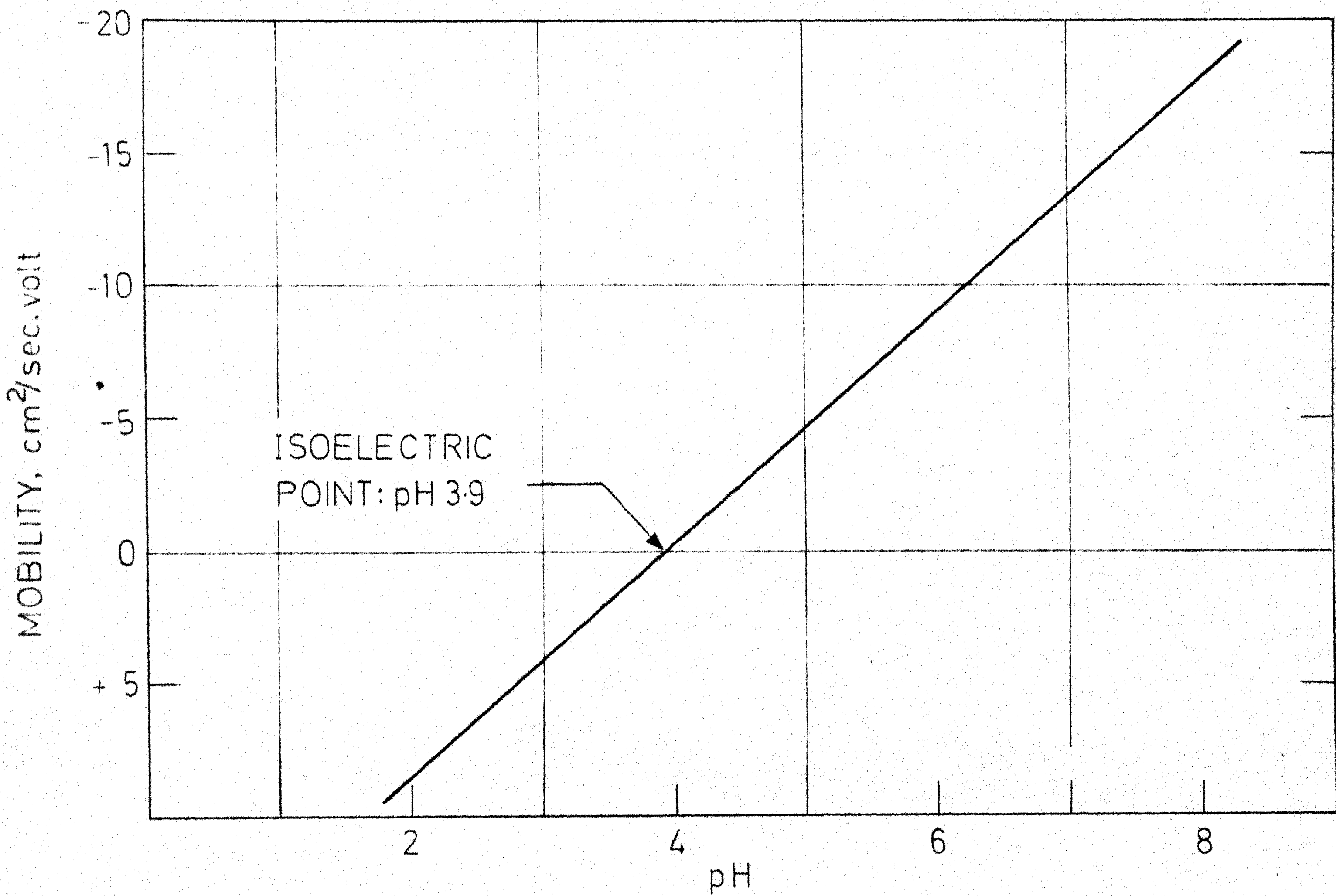

FIG. 10 ELECTROPHORETIC MOBILITY OF MS2 PHAGE

(After Overby et al. 1966)

the mobility was less in tap water and further reduced on addition of salts. Therefore, the zeta potentials determined in a particular water cannot be expected to be the same in another and zeta potentials reported should be accompanied with the ion concentration of the water. However, the change in zeta potentials with ion concentration becomes small at high concentrations.

Zeta potentials of the particles in turbid water (Table 14) were computed from electrophoretic mobility data. A precision microscope fitted with micrometer was employed for determining size of these particles.

TABLE 14

ZETA POTENTIALS OF CLAY SUSPENSIONS

Particle Size, µm	Fluid Medium	Zeta Potential, mV
0.5 to 2.0	Tube Well Water	-19
2.0 to 5.0	Tube Well Water + 7 to 8 mg/l alum (coagulated and settled)	-16

6. RESULTS AND DISCUSSION - II

The experimental results which are of relevance from a practical point of view are presented and discussed in this chapter. A discussion on dominant mechanisms of virus removal by filtration will follow in the next chapter.

6.1 Batch Sorption Tests

6.1.1 Sorption Kinetics

Batch sorption tests (non-flow agitated) were conducted to study the difference in the virus sorption potential of sand and coal. Fig. 11 shows the kinetics of sorption of MS2 phage on sand (0.5 mm) and coal (1.0 mm) at pH values 6.0, 6.8 and 8.4. The rationale for using such a range is that pH values below 7.0 are not uncommon for some waters following prechlorination or coagulation. All curves attained plateau in about 90 min and most of the sorption occurred within about 60 min. At all the three pH values, 1.0 mm coal adsorbed more number of viruses than 0.5 mm sand. For coal, virus sorption at pH 6.0 ($7.85 \times 10^3 \times 0.95 = 7.46 \times 10^3$ PFU/ml) is comparable to that at pH 6.8 ($9.5 \times 10^3 \times 0.80 = 7.6 \times 10^3$ PFU/ml). However, the corresponding values for sand are 7.07×10^3 PFU/ml ($7.85 \times 10^3 \times 0.90$) and 5.7×10^3 PFU/ml ($9.5 \times 10^3 \times 0.60$). For both sand and coal, virus sorption decreased at pH 8.4. Streaming potential measurements indicated high electronegativity (zeta potential) of sand and

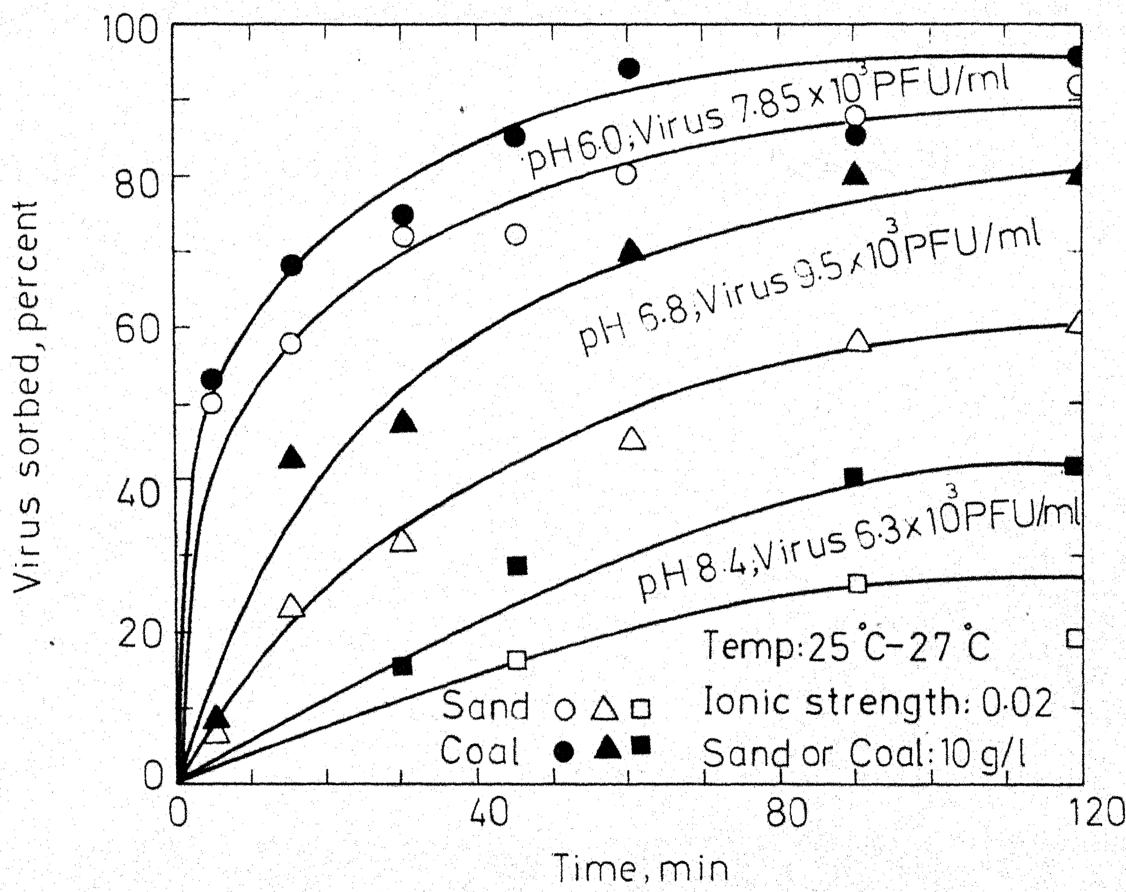

FIG. 11 KINETICS OF SORPTION OF MS2 PHAGE ON SAND AND COAL

coal at pH 8.4 and a value close to zero (slightly electronegative) at pH 6.0 and 6.8. This and high electronegativity of MS2 phage at pH 8.4 (Fig. 10) explains lower virus sorption at pH 8.4. Similarly, comparable virus sorption on coal at pH 6.0 and 6.8 can also be explained. However, the reason for the difference in virus sorption on sand at pH 6.0 and 6.8 is not readily apparent.

6.1.2 Attachment on Sand and Coal Surfaces

In order to investigate the higher virus sorption potential of coal, infrared (IR) spectra of the sand and the coal were taken using Perkin Elmer Model 137 (Perkin Elmer Corp., Norwalk, Conn., U.S.A.). Most organic and inorganic materials show absorption in the IR region and in all but a few cases this includes several characteristic wavelengths. Infrared spectra of sand and coal over a range 4000 to 650 cm^{-1} are shown in Figs. 12 and 13. Large background absorption by sand and coal precluded sharp well defined spectra. The absorption peaks at 3000, 1450, 1350 to 1375, and 700 cm^{-1} are due to Nujal base. The sand spectra show the following peaks: (i) 2350 cm^{-1}: phosphorus (-PH), (ii) 1600 to 1700 cm^{-1}: covalent nitrite (R-O-NO), covalent nitrate ($R-O-NO_2$), and imino carbonate [HN = $C(O-R)_2(s)$], (iii) 1050 to 1100 cm^{-1}: $\overleftarrow{Si}-\overrightarrow{O}-\overrightarrow{Si}$ valence stretching vibration (Simon and McMahon, 1953), (iv) 770 to

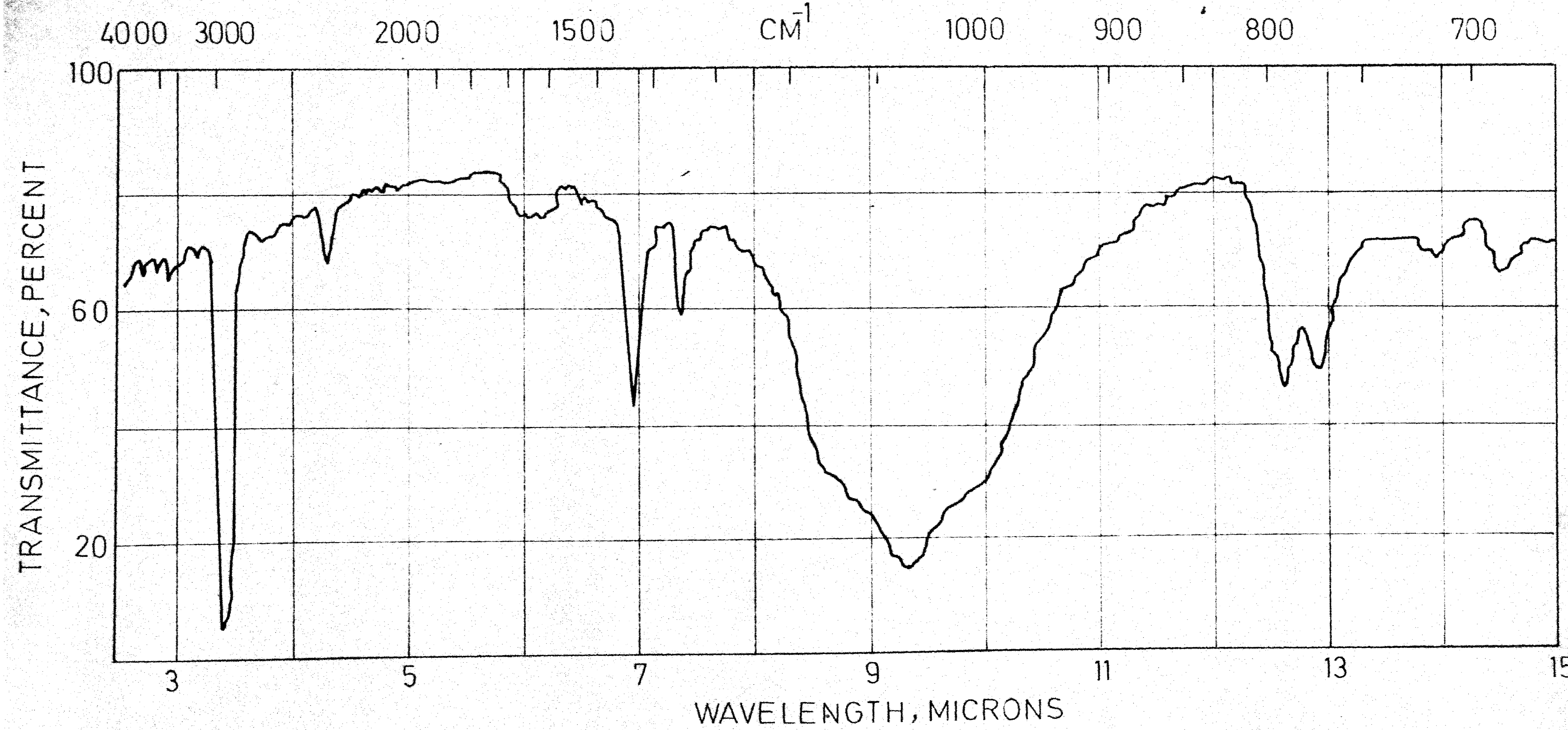

FIG. 12 INFRARED SPECTRA OF SAND

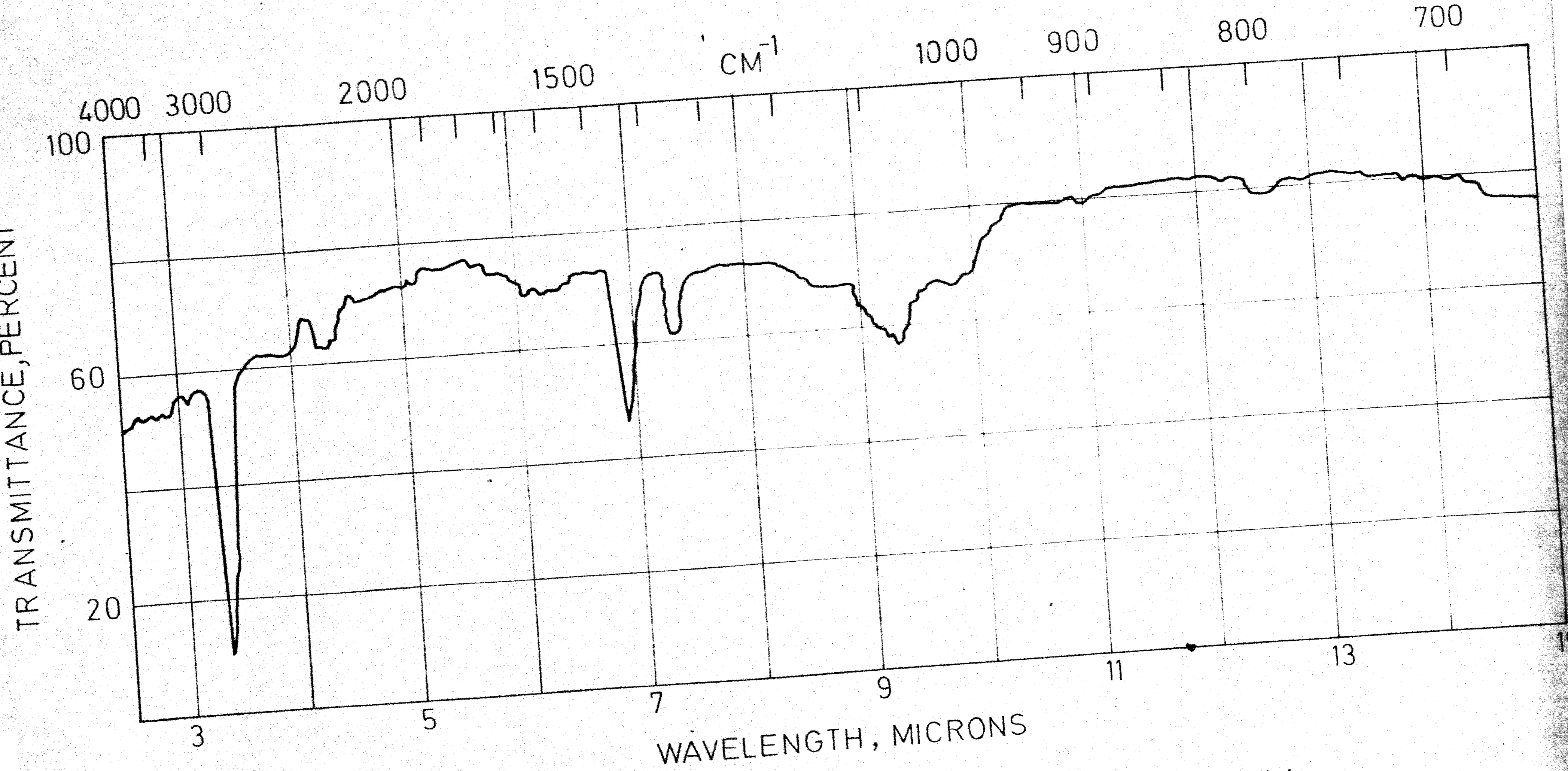

FIG. 13 INFRARED SPECTRA OF COAL

800 cm^{-1}: covalent carbonate and phosphate, and (v) 690 cm^{-1}: Si-C bond. Since only two kinds of surface groups, viz., the more reactive silanol and the less active siloxane are possible on pure silica, the other surface groups on the sand is presumably due to the impurities on the sand surface. Among the surface groups on coal, a host of those with oxygen, sulphur, halogens, and hydrogen are important. The coal spectra show the following peaks: (i) 2350 cm^{-1}: phosphorous (-PH), (ii) 1075 cm^{-1}: C-O stretching of alcohols, linear and cyclic ethers (m) (Brown, 1955), (iii) 900 cm^{-1}: alkanes such as tertiary butyl (w) alkenes [$>$ CH-CH_2 (s)], carboxyl group [CO-OH (m)], primary amines [CH_2-NH_2(s)] and [$>$ CH-NH_2 (m)], and SH group, and (iv) 790 to 800 cm^{-1} aromatic C-H out of plane vibration frequencies observed in a number of single and condensed ring structures (Brown, 1955). It is evident that the surface functional groups on the coal are chemically more active and consequently the coal exhibits a higher virus sorption compared to the sand under otherwise identical conditions. The IR spectra of sand and coal with adsorbed viruses (Fig. 14 and 15) show that the peaks at 2350 and 900 cm^{-1} for coal disappeared whereas those at 790 to 800 cm^{-1} for coal and 2350 cm^{-1} for sand reduced in intensity due to virus sorption.

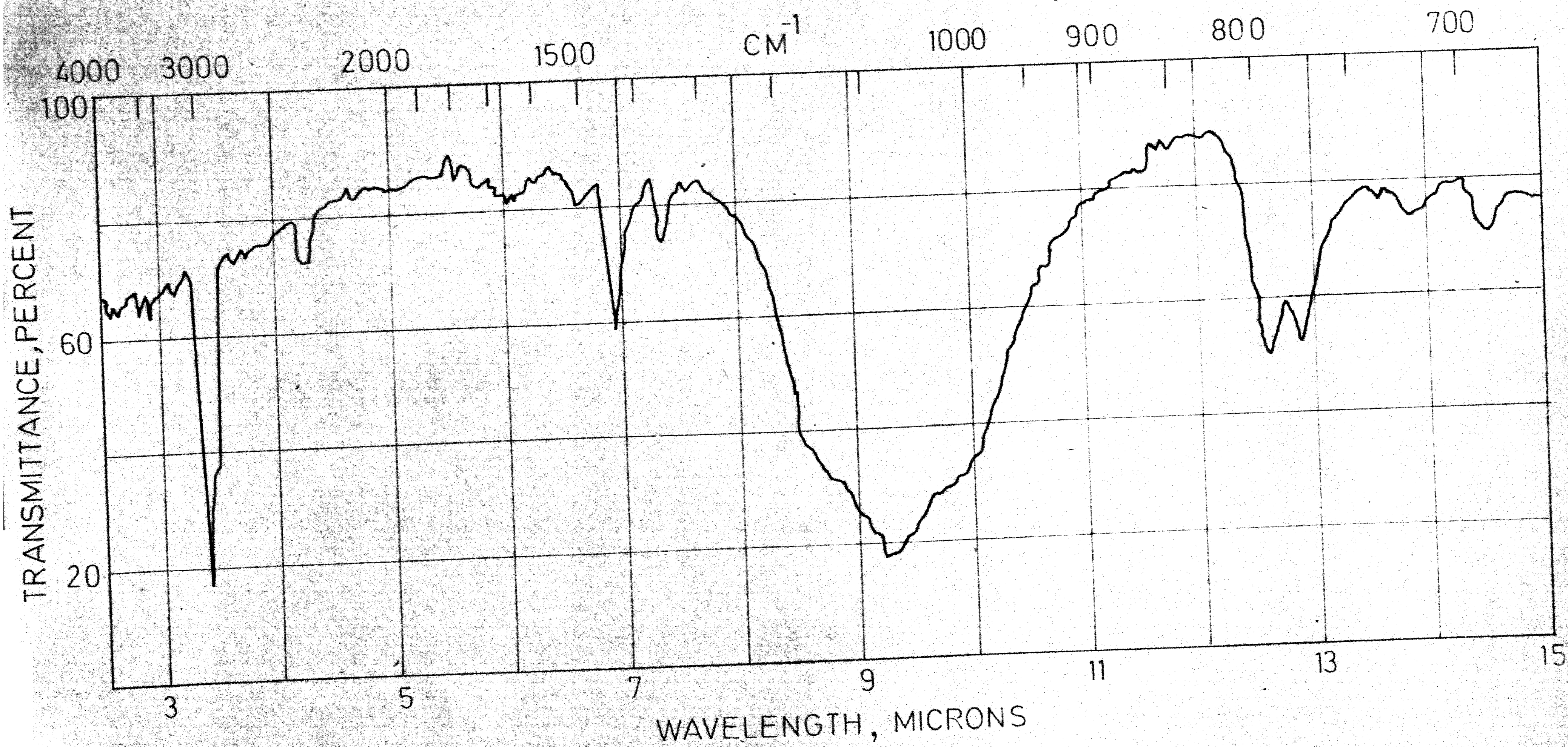

FIG.14 INFRARED SPECTRA OF SAND AFTER VIRUS SORPTION

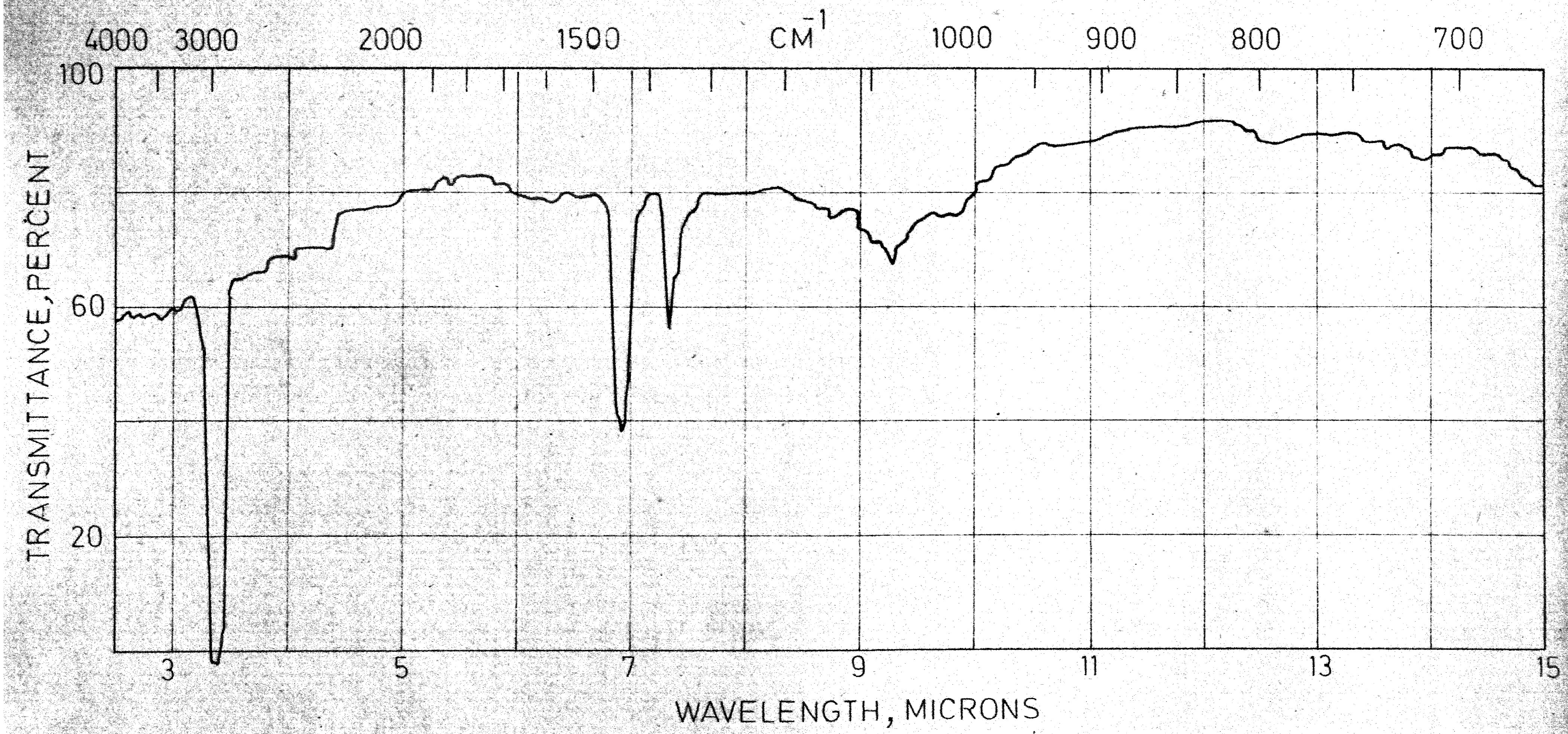

FIG.15 INFRARED SPECTRA OF COAL AFTER VIRUS SORPTION

6.2 Filtration of Discrete Virus Particles (Turbidity-Free Water)

In these experiments, whenever possible, the parameters affecting the transport step were varied while keeping the attachment conditions unaltered and vice versa. Experiments were conducted at different filtration rates (4.9, 9.8 and 14.7 m/hr) and under three different electrokinetic conditions viz., untreated bed and tube well water, untreated bed and tube well water with 0.475 meq/l Ca^{++} and alum or polyelectrolyte treated bed and tube well water. Majority of these experiments were conducted at room temperature of about 30°C. The temperature was about 36°C for those experiments conducted during the summer season. In certain experiments the pH of the influent water was altered by adding hydrochloric acid. This series of experiments demonstrates the influence of d_m, V_o and μ, parameters that affect the transport step. Also, zeta potentials of media (ζ_m) and particles (ζ_p) which influence the attachment step were varied in these experiments. The following general observations can be made from this series of experiments (Fig. 16 to 21):

a. virus removal occurs throughout the entire filter depth (Fig. 16 to 21),

b. most of the virus particles pass through untreated 0.5 mm sand or 1 mm coal filter of about 45 cm (1.5 ft) depth at 4.9 m/hr filtration rate (Fig. 17),

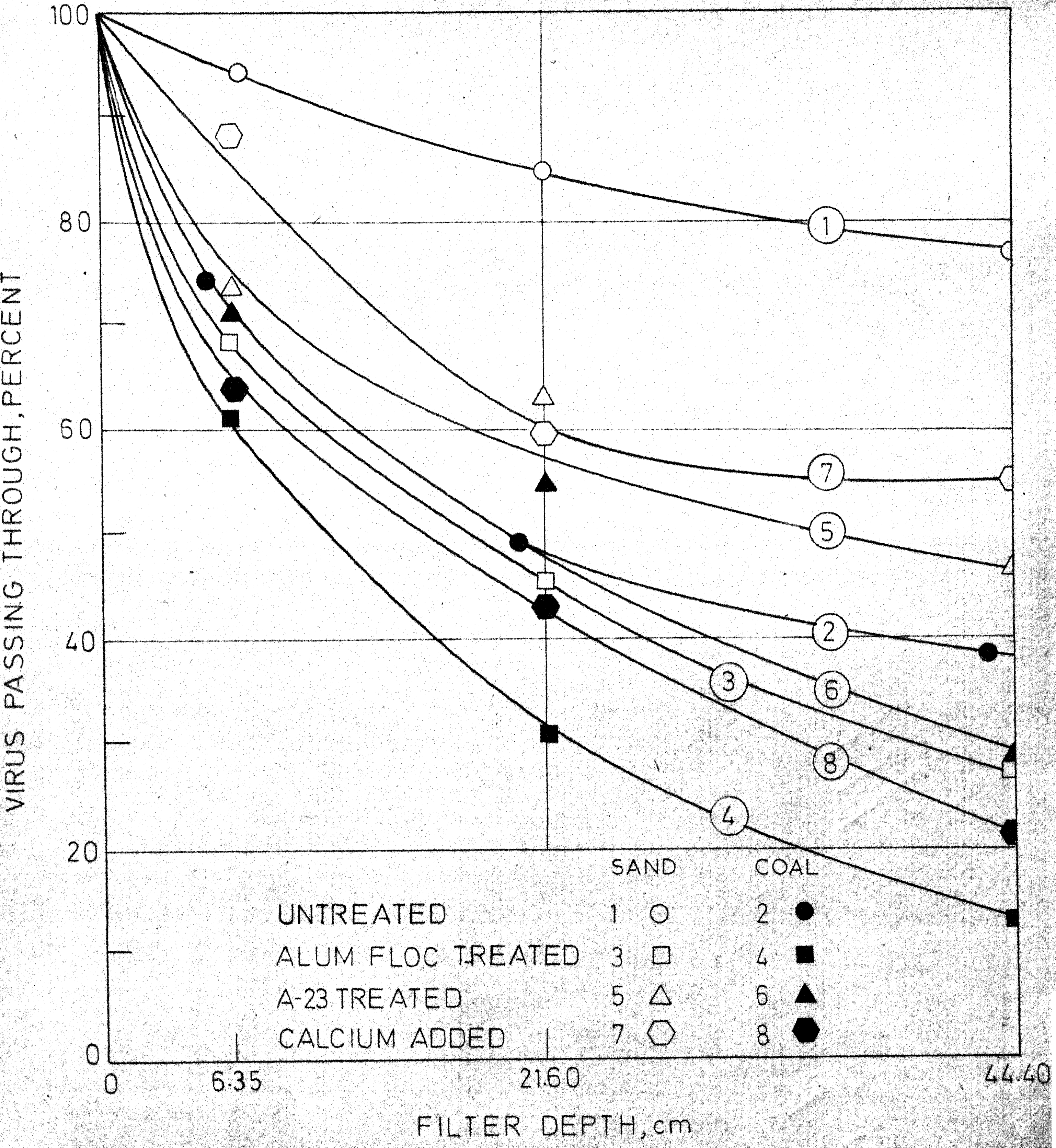

FIG.16 FILTRATION THROUGH 0.5mm SAND AND COAL UNDER VARIOUS CONDITIONS

(Ref: Tables A9, A10, A12, A13, A15, A16, A19, A20)

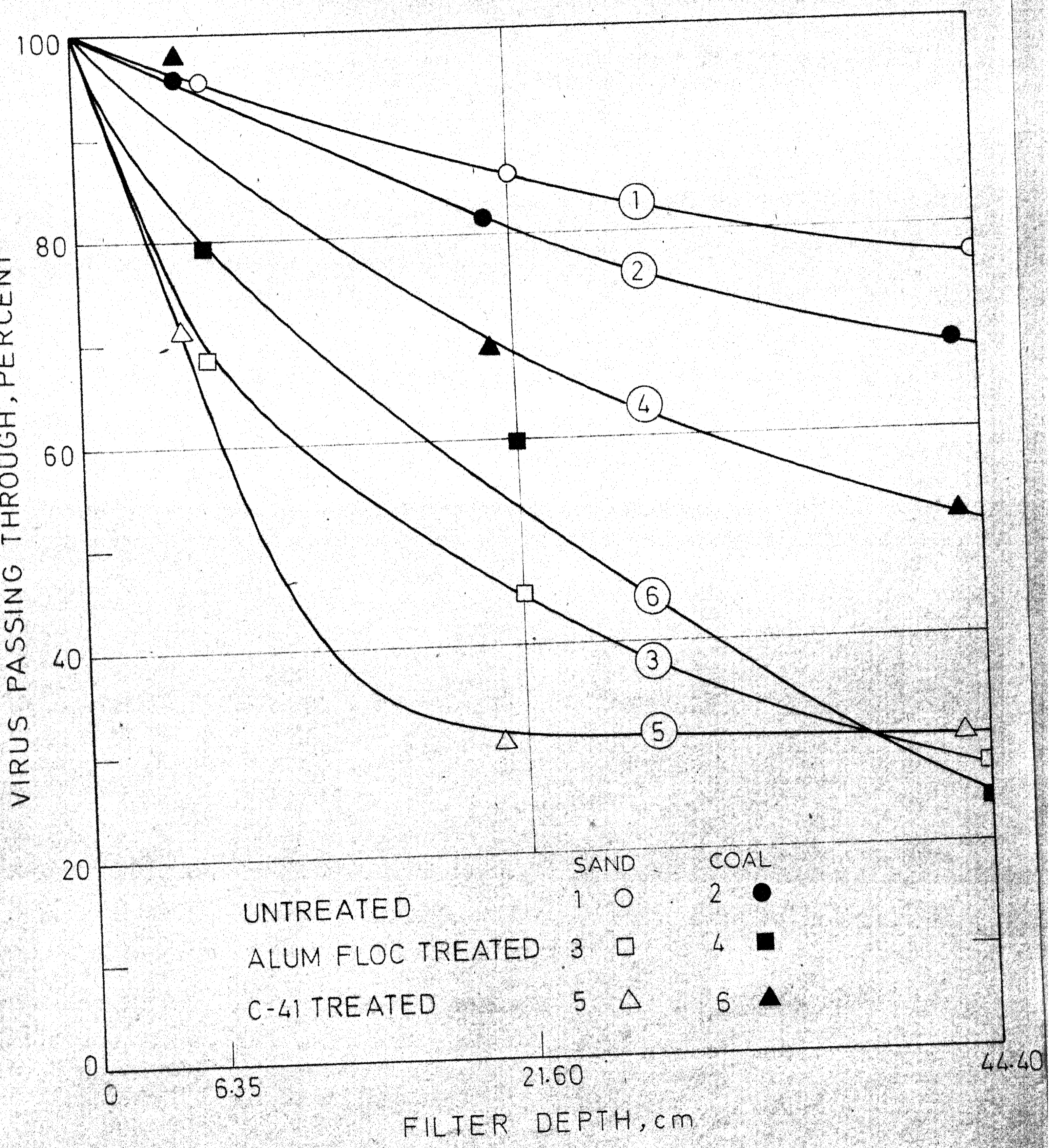

FIG. 17 FILTRATION THROUGH 0.5 mm SAND AND 1.0 mm COAL UNTREATED AND TREATED

(Ref: Tables A 9, A11, A12, A14, A17, A18)

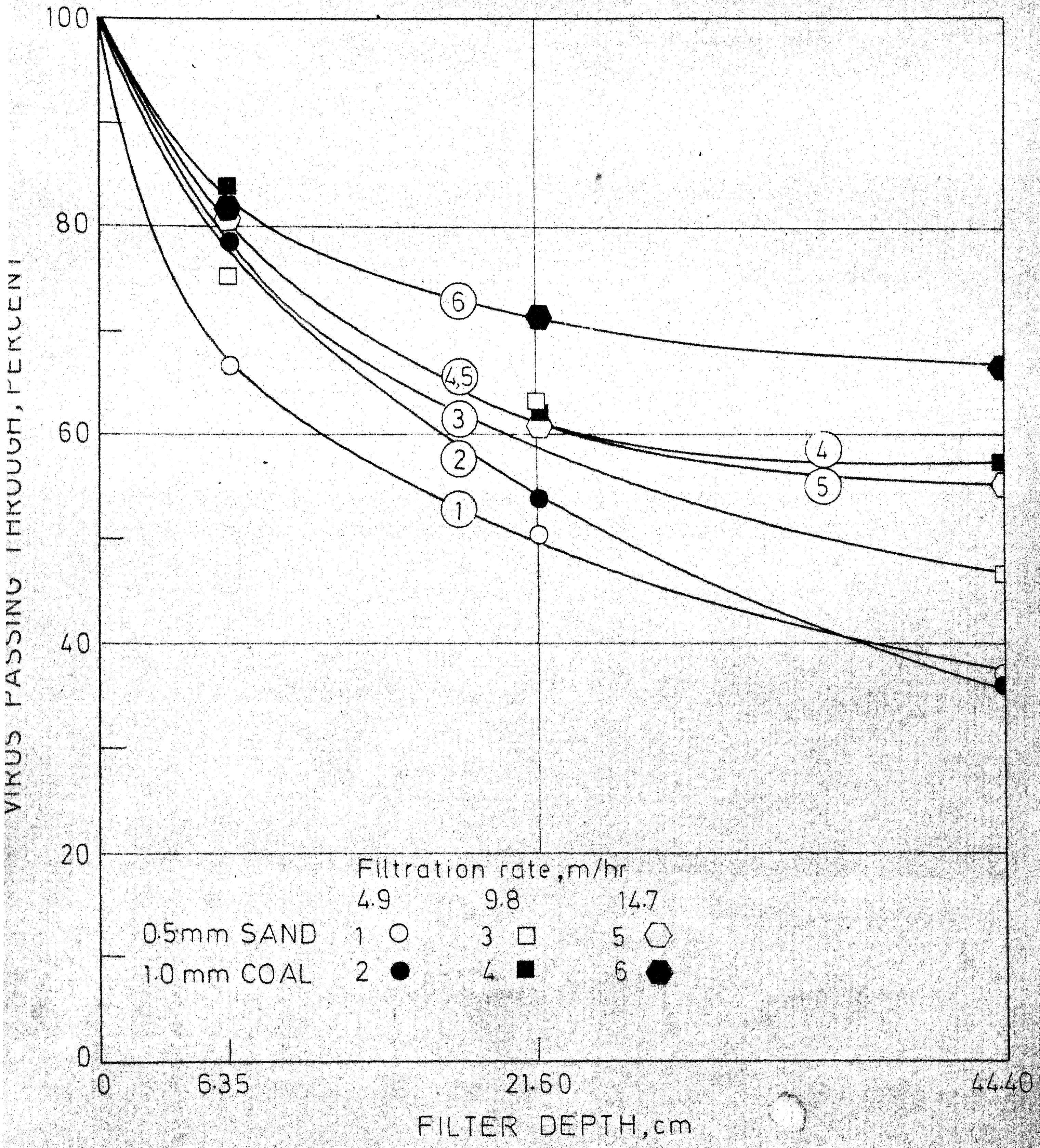

FIG. 18 EFFECT OF RATE OF FILTRATION ON FILTER PERFORMANCE (Ref: Tables A21-A26)

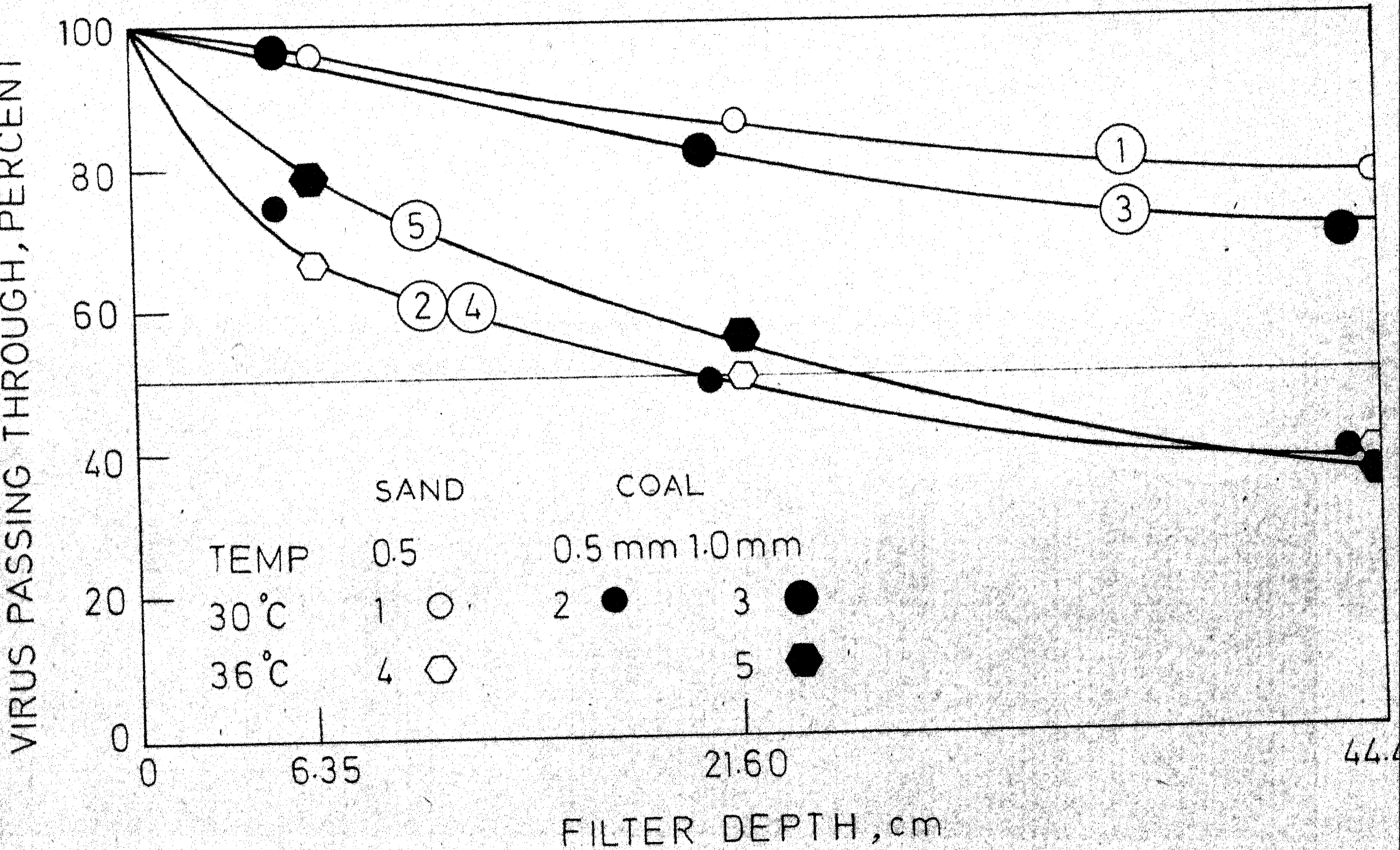

FIG. 19 EFFECT OF TEMPERATURE ON FILTER PERFORMANCE (Ref: Tables A9-A11, A21, A22)

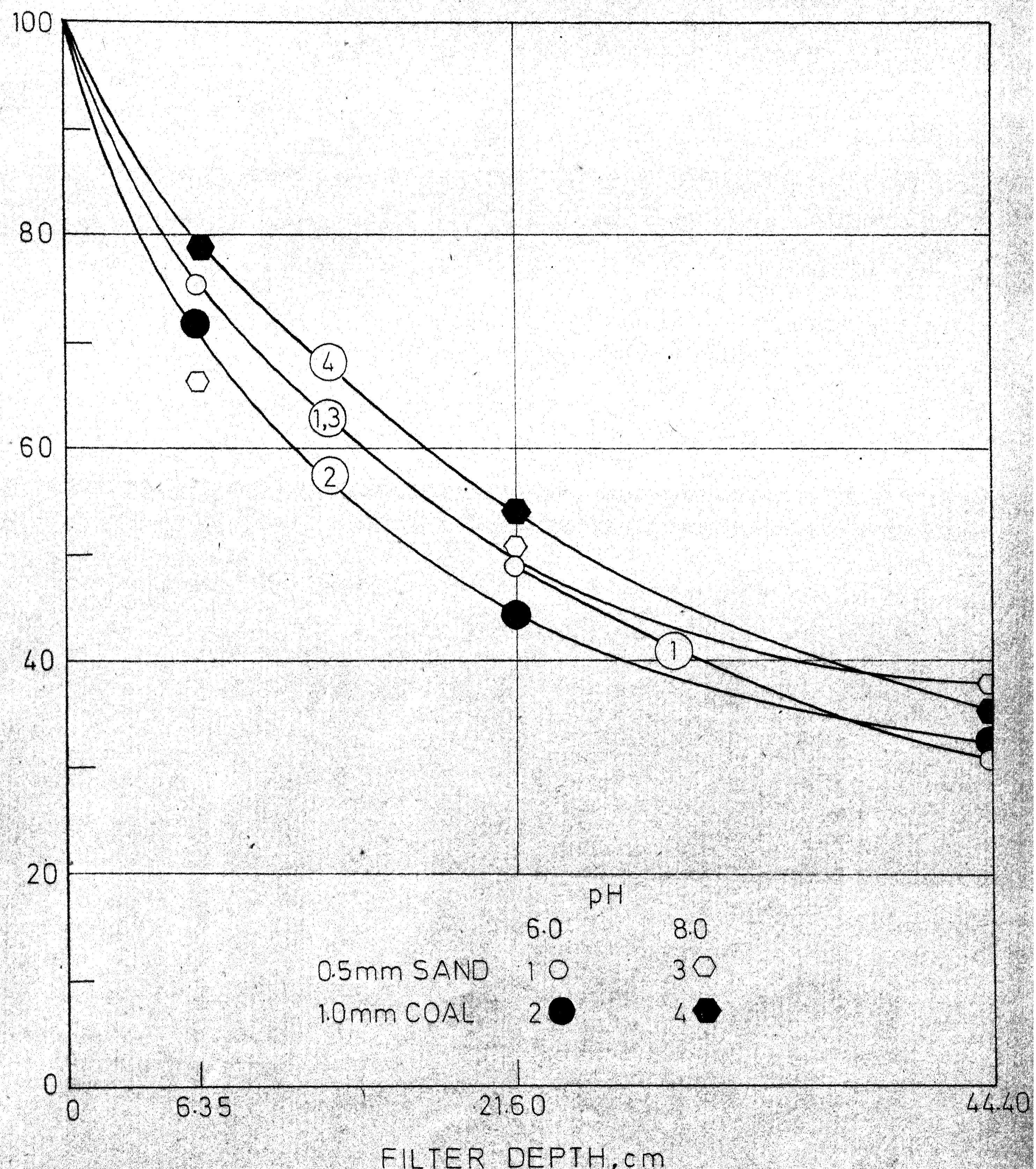

FIG.20 EFFECT OF pH ON FILTER PERFORMANCE

(Ref: Tables A21, A22, A27, A28)

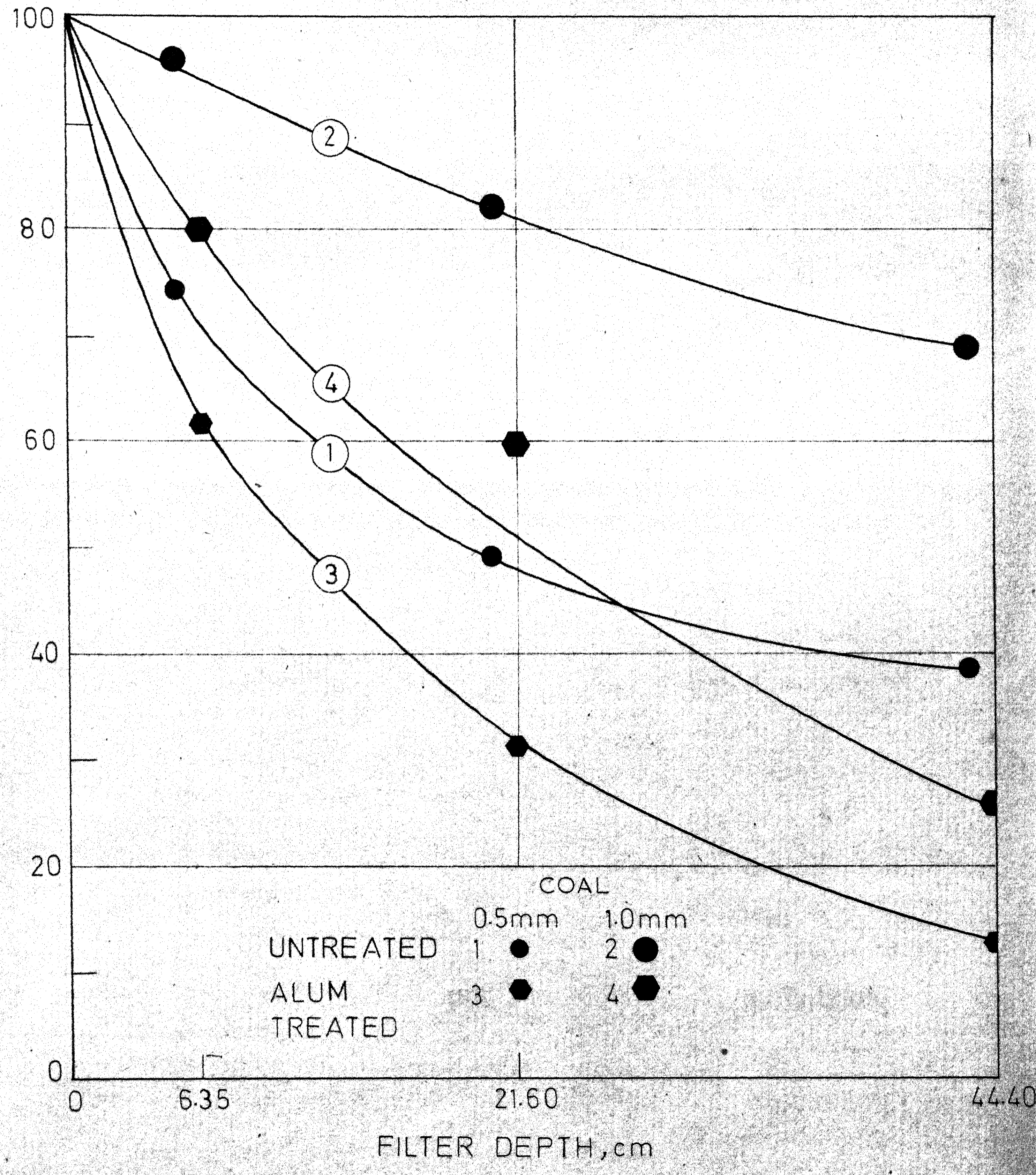

FIG.21 EFFECT OF SPECIFIC SURFACE ON FILTER PERFORMANCE (Ref: Tables A10, A11, A13, A14)

c. coal filter (untreated) of 0.5 mm size is about twice as efficient as sand filter (untreated) of same size or coal filter (untreated) of 1 mm size at a filtration rate of 4.9 m/hr (Fig. 16 to 21),

d. virus removal efficiencies of untreated coal and sand filters improve significantly by the addition of Ca^{++} to water (Fig. 16),

e. treatment with alum or cationic polyelectrolyte significantly improves the virus removal efficiencies of coal and sand filters whereas improvement due to anionic polyelectrolyte treatment is less pronounced (Fig. 16 and 17),

f. virus removal efficiencies of coal and sand filters increase with decrease in filtration rate and pH value of influent, and increase in temperature (Fig. 18 to 20).

These filtration studies were conducted for a duration of 6 to 8 hr. However, the first hour data (representing clean bed condition) are shown in Fig. 16 to 21 in which the zero hour influent virus concentrations were used in computing percent virus passing through. It was observed that the virus removal efficiencies did not deteriorate with time except in the case of alum floc treated filters.

The values of filter coefficient for different layers of a clean filter (λ_o) are presented in Table 15.

TABLE 15

FILTER COEFFICIENTS (λ_o) FOR DISCRETE VIRUS FILTRATION THROUGH SAND AND COAL

Run No.	Water	Media	Media Diameter mm	Filtration Rate m/hr	λ_o, φ cm^{-1}		
					0-6.35 cm (0-2.5 in.)	6.35-21.60 cm (2.5-8.5 in.)	21.60-44.40 cm (8.5-17.5 in.)
					$X10^{-3}$	$X10^{-3}$	$X10^{-3}$
1	Tube Well Water	Untreated Sand	0.5	4.9	7.4	7.6	4.2
2	"	Untreated Coal	0.5	4.9	57.7	27.8	10.5
3	"	Untreated Coal	1.0	4.9	8.1	10.2	7.5
4	Tube Well Water + 0.475 meq/l Ca^{++}	Untreated Sand	0.5	4.9	19.6	25.8	3.5
5	"	Untreated Coal	0.5	4.9	70.1	26.0	30.3
6	Tube Well Water	Sand Treated With C-41	0.5	4.9	66.3	54.5	0.9
7	"	Coal Treated With C-41	0.5	4.9	18.8	18.2	12.2
8	"	Sand Treated With Alum	0.5	4.9	59.2	28.0	24.5

Tube Well Water; pH ≈ 8 (range 7.8 to 8.2) for Run No. 1 to 18; Temp. ≈ 30°C (range 28°C to 31°C) for Run No. 1 to 12, ≈ 36°C (range 35°C to 37°C) for Run No. 13 to 21.

φ $\lambda_o = \frac{1}{L} \text{Ln} \frac{C_1}{C_2}$ where, L = depth of layer, and C_1, C_2 = virus concentration at the top and bottom of layer, respectively.

(Data Source: Tables A9 to A29).

TABLE 15 (Contd.)

Run No.	Water	Media	Media Diameter mm	Filtration Rate m/hr	λ_o, φ cm^{-1}: 0-6.35 cm (0-2.5 in.)	6.35-21.60 cm (2.5-8.5 in.)	21.60-44.40 cm (8.5-17.5 in.)
9	Tube Well Water	Coal Treated With Alum	0.5	4.9	76.7	44.7	40.5
10	"	Coal Treated With Alum	1.0	4.9	35.8	30.7	30.6
11	"	Sand Treated With A-23	0.5	4.9	47.3	10.3	13.3
12	"	Coal Treated With A-23	0.5	4.9	67.4	26.6	22.3
13	"	Untreated Sand	0.5	4.9	64.4	17.9	13.5
14	"	Untreated Coal	1.0	4.9	38.0	24.4	17.7
15	"	Untreated Sand	0.5	9.8	45.0	16.5	10.3
16	"	Untreated Coal	1.0	9.8	28.1	19.8	3.3
17	"	Untreated Sand	0.5	14.7	33.6	19.0	3.8
18	"	Untreated Coal	1.0	14.7	32.9	9.2	2.7
19	Tube Well Water pH Adjusted to 6	Untreated Sand	0.5	4.9	43.9	28.1	20.5
20	"	Untreated Coal	1.0	4.9	51.8	31.5	14.0
21	Tube Well Water pH Adjusted to 7	Untreated Sand	0.5	4.9	56.7	22.1	21.2

From these values of λ_o and the information from Fig. 16 to 21, the effects of d_m, V_o, μ, ζ_m and ζ_p on filter performance can be better correlated.

6.2.1 Effect of Diameter of Filter Media

According to the empirical relationship proposed by Ives on the basis of plain sand filtration of 1.3 µm latex particles (Equation 6), λ_o varies inversely with diameter of filter media. Based on microscopic observations, it has been reported in the literature that the pore size in a granular filter is of the order of the grain size. Therefore, the smaller the diameter of the filter grain smaller the distance that the particles are to be transported. Further, the specific surface of a filter and hence the available sites for attachment increase with decrease in grain diameter. Therefore, Ives' correlation of the filter coefficient (λ_o) with the inverse of media diameter has theoretical justification from the viewpoints of both transport and attachment steps in filtration. However, Agrawal (1966) observed that this correlation did not hold good for filtration of 1.3 µm latex particles through cationic polyelectrolyte treated sand. It is rather difficult to precisely correlate λ_o with filter media diameter on the basis of limited data available in the present study. The λ_o ratios for different layers of 0.5 mm

and 1 mm coal filters computed from λ_o values of Table 15 are 7.12, 2.75 and 1.4 for untreated media and 2.14, 1.455 and 1.323 for media treated with alum floc.

6.2.2 Effect of Rate of Filtration

Ives' empirical equation (Equation 6) also correlates λ_o inversely with the rate of filtration. Electrokinetic removal mechanism (Equation 26) as well as the transport model due to gravity settling (Equation 12) correlate the filter coefficient inversely with the rate of flow (velocity) through a filter. However, the transport due to gravity settling is insignificant in discrete virus filtration (Table 8). The transport model due to diffusion correlates λ_o with inverse 2/3 power of the rate of filtration. The correlation of λ_o with either the inverse or inverse 2/3 power will indicate the dominant transport model. The λ_o values at different filtration rates for sand and coal filters are analyzed here (Filter Run 13 to 18, Table 15). For an inverse velocity ratio of 2 (4.9 and 9.8 m/hr), the λ_o ratios for the consecutive layers of the sand filter are 1.432, 1.085, and 1.31 whereas those for the coal filter are 1.358, 1.232, and 5.37. Similarly, for an inverse velocity ratio of 3 (4.9 and 14.7 m/hr), the ratios are 1.92, 0.9425, and 3.55 for the sand and 1.155, 2.65, and 6.56 for the coal filters. Corresponding to an inverse velocity ratio of 1.5

(9.8 and 14.7 m/hr), the λ_o ratios are 1.34, 0.87, and 2.71 for the sand and 0.855, 2.15, and 1.222 for the coal filters. Therefore, inverse velocity ratios do not correlate well with the ratios of the filter coefficients. This rules out electrokinetic mechanism to be the only dominant mechanism for the transport of the discrete virus particles. The inverse 2/3 power velocity ratios become 1.587, 2.08, and 1.31 for 4.9 and 9.8 m/hr, 4.9 and 14.7 m/hr, and 9.8 and 14.7 m/hr filtration rates, respectively. These ratios of 1.587, 2.08 and 1.31 are very close to the corresponding filter coefficient ratios 1.432, 1.92 and 1.34 for only the top layer of the sand filter whereas such a relationship is not observed for any of the layers of the coal filter. This suggests that diffusion also is not the only dominant transport mechanism. Probably both diffusion as well as electrokinetic phenomena lead to effective transport in discrete virus filtration. This analysis, of course, holds good provided the attachment step is not limiting.

6.2.3 Effect of Temperature

According to the literature on filtration reviewed by Agrawal (1966), better filter performance is generally observed during summer months indicating that temperature is an important parameter in the removal of suspended particles by granular filtration. However,

satisfactory explanation for such an observation was not offered by many investigators. A scrutiny of the transport models reveals that an increase in temperature greatly enhances transport due to diffusion (Table 8). This is because a rise in temperature increases the numerator of the diffusion model and decreases the denominator as the viscosity reduces at higher temperatures. Equation 12 correlates the transport due to gravity settling and viscosity inversely, i.e. for an increase in temperature the resulting reduced viscosity will improve the transport of particles. Further, a change in temperature may also change the ionization of the media as well as the suspended particles which may result in improved attachment conditions. The electrokinetic removal mechanism (Equation 26) also responds to changes in temperature. Of the filtration mechanisms effected by temperature, gravity settling is insignificant in discrete virus filtration.

Fig. 19 shows virus removal by sand and coal filters at temperatures of 30°C and 36°C at filtration rate of 4.9 m/hr. Filtration through 0.5 mm sand and 1 mm coal produced 23 and 63, and 31 and 64 percent virus removals at 30°C and 36°C, respectively. According to the Metropolitan Water Board, London (1971-73), a slow sand filter operating at a standard rate of 4.8 m/day at a temperature of 11°C to 12°C reduced poliovirus 1 by 99.999 percent whereas the

same filter removed 99.8 percent at $6^{o}C$, a hundred fold difference. Eventhough this hundred fold difference was attributed to difference in temperature, it is probable that a part of this could have been due to the difference in biological activity in the schmutzdecke at these two temperatures. Nevertheless, it is evident that temperature is an important parameter in filtration and presumably it effects both the transport and attachment of suspended particles. However, it is to be noted that this parameter cannot be economically altered to advantage in the treatment of water by filtration.

6.2.4 Effect of Zeta Potentials

Significant improvement in virus removal was observed in the case of untreated media when Ca^{++} was added to the influent water or when the filter media was coated with either alum floc or polyelectrolyte (Fig. 16, 17 and Table 15). This improvement in virus removal is presumably due to a change in the zeta potentials of either filter media or filter media and the virus particles and consequently, an overall improvement in the attachment conditions. The zeta potentials of untreated and treated filter media are shown in Table 13. When Ca^{++} is added to water the zeta potentials of both the suspended particles and filter media are likely to change and become either less negative or

positive. The zeta potentials of sand and coal media for various doses of Ca^{++} are shown in Table 12. However, the zeta potential of the discrete virus particles could not be determined during this investigation as mentioned earlier. Studies by Agrawal (1966) indicated that by adding 1 m<u>M</u> Ca^{++} to water the zeta potentials of various particles became less negative [1.3 µm latex particles: -27 mV to -21 mV, algae (<u>Scenedesmus quadripoda</u>): -45 mV to -29 mV, <u>Escherichia coli</u>: -33 mV to -21 mV, and bentonite clay: -26 mV to -19 mV] whereas addition of a similar dose of trivalent cations (aluminum) reversed the sign of zeta potential. Assuming that even on addition of Ca^{++} the discrete virus particles have a negative zeta potential, the improvement in virus removal in the case of **treated** media or untreated media with 0.475 meq/l Ca^{++} could be due to unlike zeta potentials of filter media and the virus particles. It was observed that when the filter media was treated with an anionic polyelectrolyte, eventhough the negative zeta potentials were increased from -18.5 mV to -35.2 mV in case of sand and from -23.1 mV to -33.8 mV in case of coal, the virus removals were higher compared to filtration through untreated media (Fig. 16). This improvement in virus removal could mainly be attributed to the adsorption of viruses to the **treated** filter grains through H^{+} and Ca^{++} bridging (Ives, 1971, and Carlson, <u>et al</u>., 1968).

6.2.5 Effect of pH

O'Melia and Crapps (1964) reported that efficiency of filtration was poorest at pH 9.5 and attributed this to strong negative mobilities exhibited by both the suspended particles and the sand media. This agrees with the observation made by Stanley (1955) that filtration became more difficult as the pH level increased above 7. O'Melia and Crapps also observed that the surface properties of clean sand varied with pH, and the type and valence of the cations in the water. O'Melia and Stumm (1967) recognized pH to be one of the chemical parameters which could influence particle attachment. Further, they reported that no unique relationship existed between pH and filtration efficiency due to considerable overlapping by other chemical parameters. Therefore, it is probable that pH effects attachment most probably through its influence on the ionization of surface groups on both filter media and suspended particles. Fig. 20 shows filter performance at pH 6.0 and 8.0. The virus removals increased from 63 to 69 percent for 0.5 mm sand and 65 to 68 percent for 1 mm coal filters at 36°C when the pH was adjusted from 8 to 6.

6.2.6 Variation in λ_o Along the Depth of Filter

The filter coefficient (λ_o) varies for different filter systems depending upon the dominant transport and

attachment conditions. Within a filter, a variation in λ_o through different layers may be due to either size distribution of suspended particles or stratification of filter media (due to backwashing) making some of the mechanisms operative to different degrees along the depth of the filter, or due to the effect of decreasing concentration of suspended particles along the depth of the filter as predicted by the electrokinetic removal mechanism (Equation 26). For filtration of discrete virus particles through an unstratified filter bed of fairly close media size range used in this study, the first two reasons do not hold good. However, in all filter runs the values of λ_o decreased from top to bottom of the filter (Table 15) and it seems, therefore, that the electrokinetic mechanism is a significant mechanism in virus removal by filtration. Also, the percent variation in the λ_o values from top to bottom layers of the filter is much less for alum or polyelectrolyte treated sand and coal compared to the untreated media. This further supports the importance of electrokinetic removal mechanism in virus filtration as the second term of Equation 26, which is independent of the concentration of particles becomes additive when the filter bed is treated with polyelectrolyte or alum floc rendering the media zeta potential positive.

Further, it was observed that the percent variation in λ_o values from top to bottom of the filter is

much less in the case of untreated coal compared to untreated sand at a normal filtration rate of 4.9 m/hr (Table 15). This implies that coal filtration is effective even at low concentration of virus particles. This cannot be explained satisfactorily from the electrokinetic mechanism of removal alone and leads to the belief that coal surface has higher attachment potential.

6.2.7 Sand and Coal as Filter Media

An examination of Fig. 16 to 20, and Tables 15 and 16 reveals that filtration through coal media results in far better removals of virus particles than sand media of the same size at same filtration rate. In filtration through sand or coal of same size, the transport conditions are identical. The removals due to electrokinetic phenomena will also be of the same order or slightly better with sand media because of the slightly higher negative zeta potential of coal. But the observed removals are contrary to this. Further, it was observed that the filter coefficient λ_o is almost same for 0.5 mm sand and 1 mm coal media at 4.9 m/hr flow rate. The specific surface provided by 1 mm coal is about half that provided by 0.5 mm sand. In addition, the suspended particles are to be transported over comparatively greater distance from the bulk flow near the media surface in the case of 1 mm coal compared to

TABLE 16

SAND AND COAL AS FILTER MEDIA UNDER SIMILAR CONDITIONS OF OPERATION (DISCRETE VIRUS FILTRATION)

Conditions of Filter Operation	0.5 mm Sand		1 mm Coal		0.5 mm Coal		Probable Reason for the Difference	Remarks
	Percent Virus Removal (a)	Difference in Percent Removal φ (w.r.t.)	Percent Virus Removal (b)	Difference in Percent Removal φ (w.r.t.)	Percent Virus Removal (c)	Difference in Percent Removal φ (w.r.t.)		
Tube Well Water; pH ≈ 8; Temp. ≈ 30°C; Filtration Rate 4.9 m/hr								
1. Untreated Media (Fig. 16,17)	23		31		62			Untreated coal about 2.7 times efficient than sand of the same size. Coal surface has higher attachment potential.
2. Alum Treated Media (Fig. 16,17)	72	+49 (1a)	75	+44 (1b)	87	+25 (1c)	Improved attachment	Improvement in attachment condition improves removal greatly by sand and to a lesser extent by coal suggesting poor attachment potential of natural sand.
3. C-41 Treated Media (Fig. 17)	69	+46 (1a)	48	+17 (1b)			Improved attachment	
4. A-23 Treated Media (Fig. 16)	53	+30 (1a)			71	+9 (1c)		

φ Increase +; decrease -

TABLE 16 (Contd.)

Conditions of Filter Operation	0.5 mm Sand		1 mm Coal		0.5 mm Coal		Probable Reason for the Difference	Remarks
	Percent Virus Removal (a)	Difference in Percent Removal φ (w.r.t.)	Percent Virus Removal (b)	Difference in Percent Removal φ (w.r.t.)	Percent Virus Removal (c)	Difference in Percent Removal φ (w.r.t.)		
5. Untreated Media, Ca^{++} Added to Tube Well Water (Fig. 16)	45	+22 (1a)			79	+17 (1c)	Improved attachment	
Tube Well Water; pH ≈ 8; Temp. ≈ 36°C								
6. Untreated Media 4.9 m/hr (Fig. 18)	63	+40 (1a)	64	+33 (1b)			Improved Transport and Attachment	0.5 mm sand and 1 mm coal filters remove viruses to the same extent. Poor transport in 1 mm coal is probably balanced by poor attachment in 0.5 mm sand.

φ Increase +; decrease -

TABLE 16 (Contd.)

Conditions of Filter Operation	0.5 mm Sand		1 mm Coal		0.5 mm Sand		Probable Reason for the Difference	Remarks
	Percent Virus Removal (a)	Difference in Percent Removal φ (w.r.t.)	Percent Virus Removal (b)	Difference in Percent Removal φ (w.r.t.)	Percent Virus Removal (c)	Difference in Percent Removal φ (w.r.t.)		
7. Untreated Media 9.8 m/hr (Fig. 18)	54	-9 (6a)	42	-22 (6b)			Deteriorated Transport	Because of poor transport the decrease is more pronounced in the case of 1 mm coal.
8. Untreated Media 14.7 m/hr (Fig. 18)	44	-19 (6a)	33	-31 (6b)				

φ Increase +; decrease -

0.5 mm sand. The fact that the virus removals were of the same order in 0.5 mm sand and 1 mm coal filtration even under these two comparatively unfavourable conditions for coal demonstrates that coal surface has a much greater attachment potential. This is probably due to the predominance of chemically active surface functional groups (Section 6.1.2) which make possible more number of collisions effective.

6.3 Filtration of Virus and Clay Turbidity

In natural surface waters viruses occur along with other particulate matter causing turbidity. It was felt that filtration studies wherein viruses were present in association with clay turbidity would be of practical significance. In this series of experiments virus concentration and headloss were monitored along the depth of the filter in single-media sand (4.9 m/hr) and coal-sand dual-media (9.8 m/hr) filters. Low turbidity measurements using the Hellige turbidimeter require about 150 ml sample (50 mm viewing depth). It was felt that collection of such large volume samples for turbidity measurement would interfere with normal laminar flow in the experimental filter column and therefore, only the influent and effluent turbidities were monitored. Experiments were conducted under three different filtration conditions, viz., filtration of virus and clay turbidity through (i) untreated media (particle and

media zeta potentials negative), (ii) polyelectrolyte treated media (zeta potential of particle negative whereas that of media positive), and (iii) untreated media following coagulation and settling (particle and media zeta potentials negative but to a lesser extent compared to (i)).

The results of this series of experiments are shown in Fig. 22 to 24 from which the following general observations can be made: (i) virus removal is confined mostly to the top layers in sand and spread over the entire depth in dual-media filtration, (ii) headloss development is most rapid in the top layers of sand filter and spread over the entire depth in dual-media filter, (iii) headloss through sand at 4.9 m/hr is comparable to that through dual-media at 9.8 m/hr for filter runs upto 5.5 hr, and (iv) filter performance (virus and turbidity removal) under the three filtration conditions were either almost constant throughout the filter run or showed improvement with time. The first two observations indicate similar trend for virus and turbidity removal in sand or dual-media filter; however, the trend in each filter is different.

Table 17 presents the values of filter coefficient (λ_0) and the percent virus removal (clean bed condition) for filtration of discrete virus as well as virus in association with clay turbidity through untreated and treated media. For filtration through untreated sand, virus removal increased from 22.75 percent (discrete virus) to 73 percent

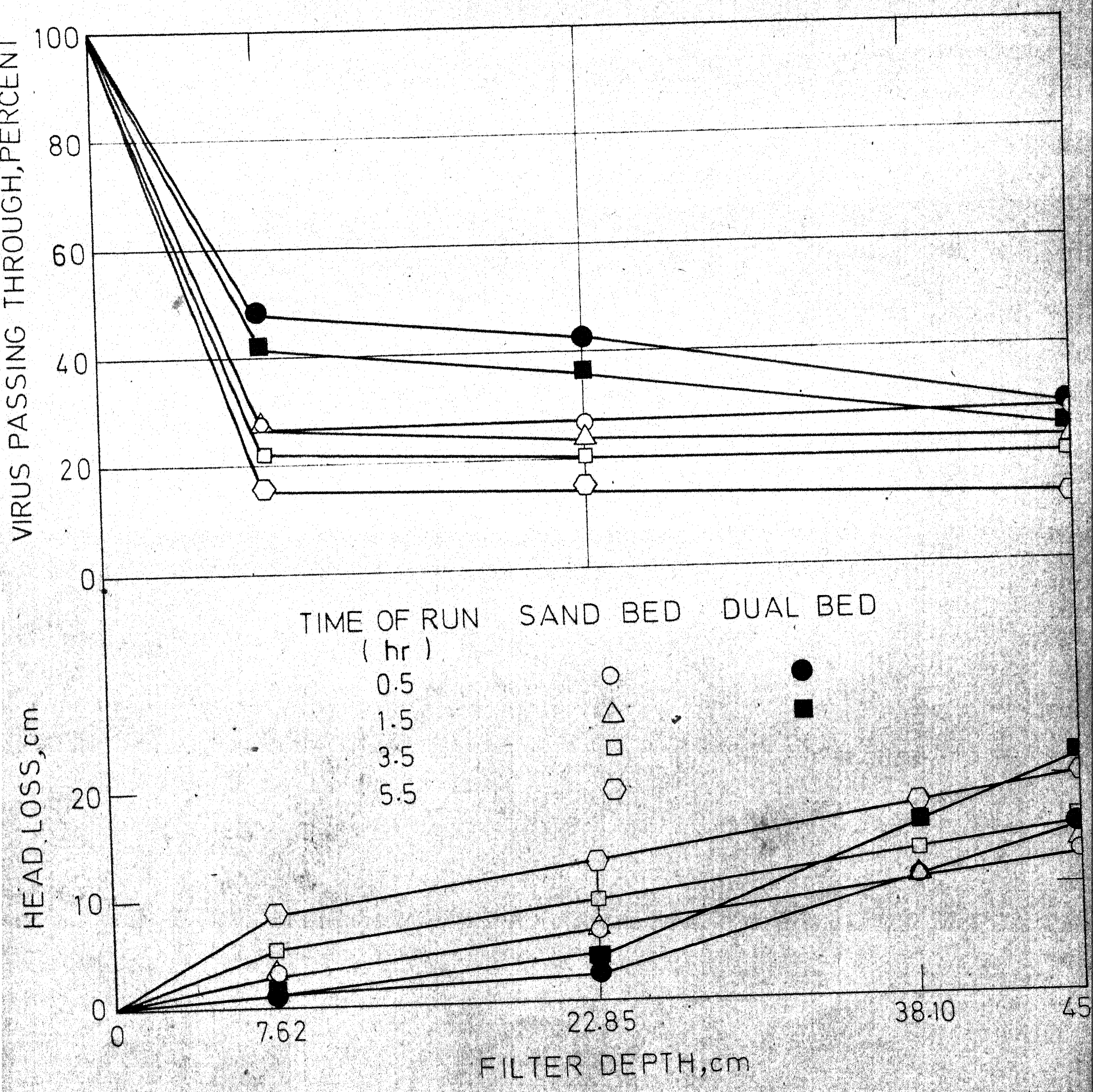

FIG.22 FILTRATION OF VIRUS AND CLAY TURBIDITY THROUGH UNTREATED SAND AND DUAL MEDIA (Ref: Tables A30, A31)

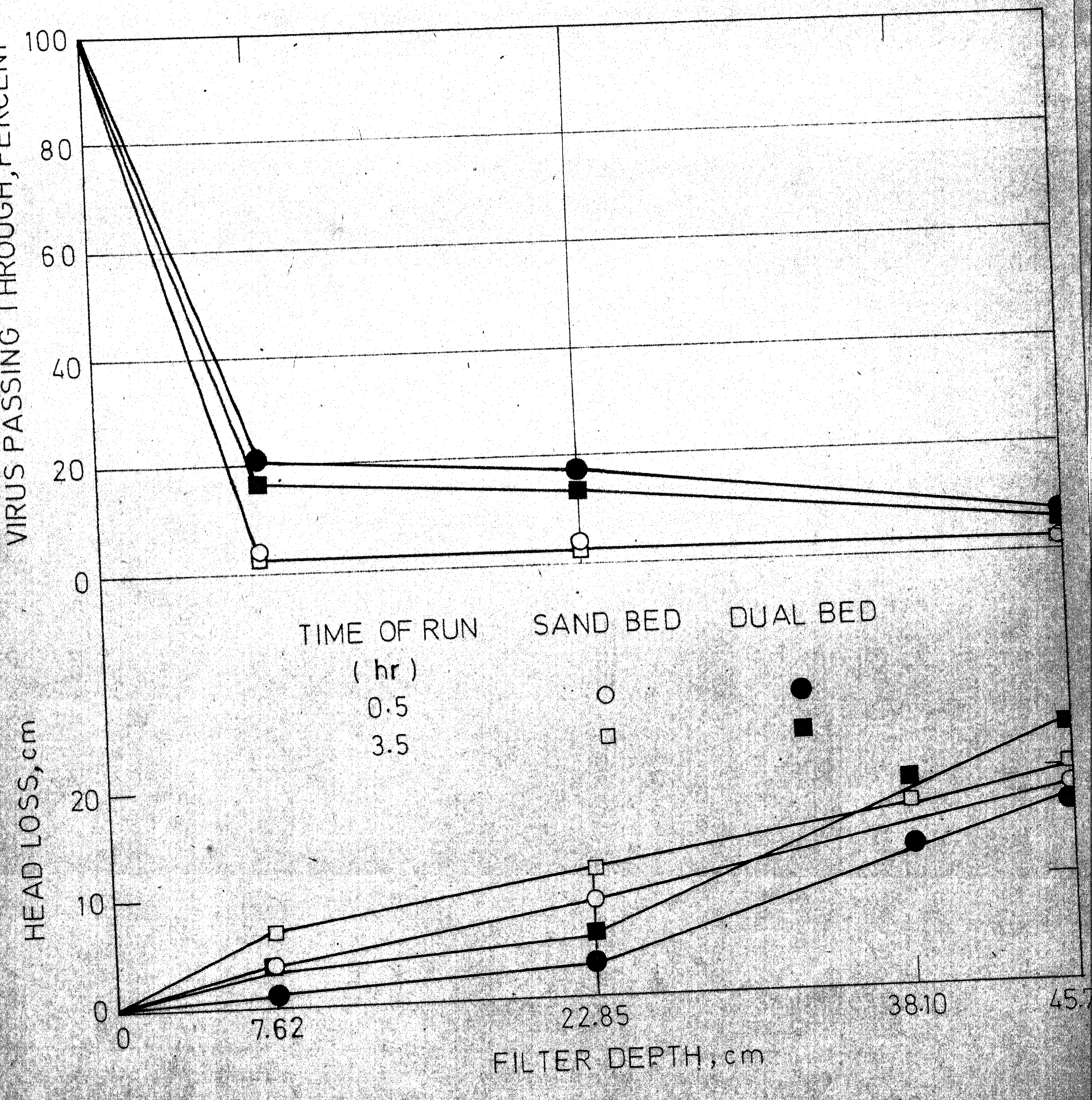

FIG.23 FILTRATION OF COAGULATED, FLOCCULATED AND SETTLED SUSPENSIONS THROUGH SAND AND DUAL MEDIA (Ref: Tables A30, A31)

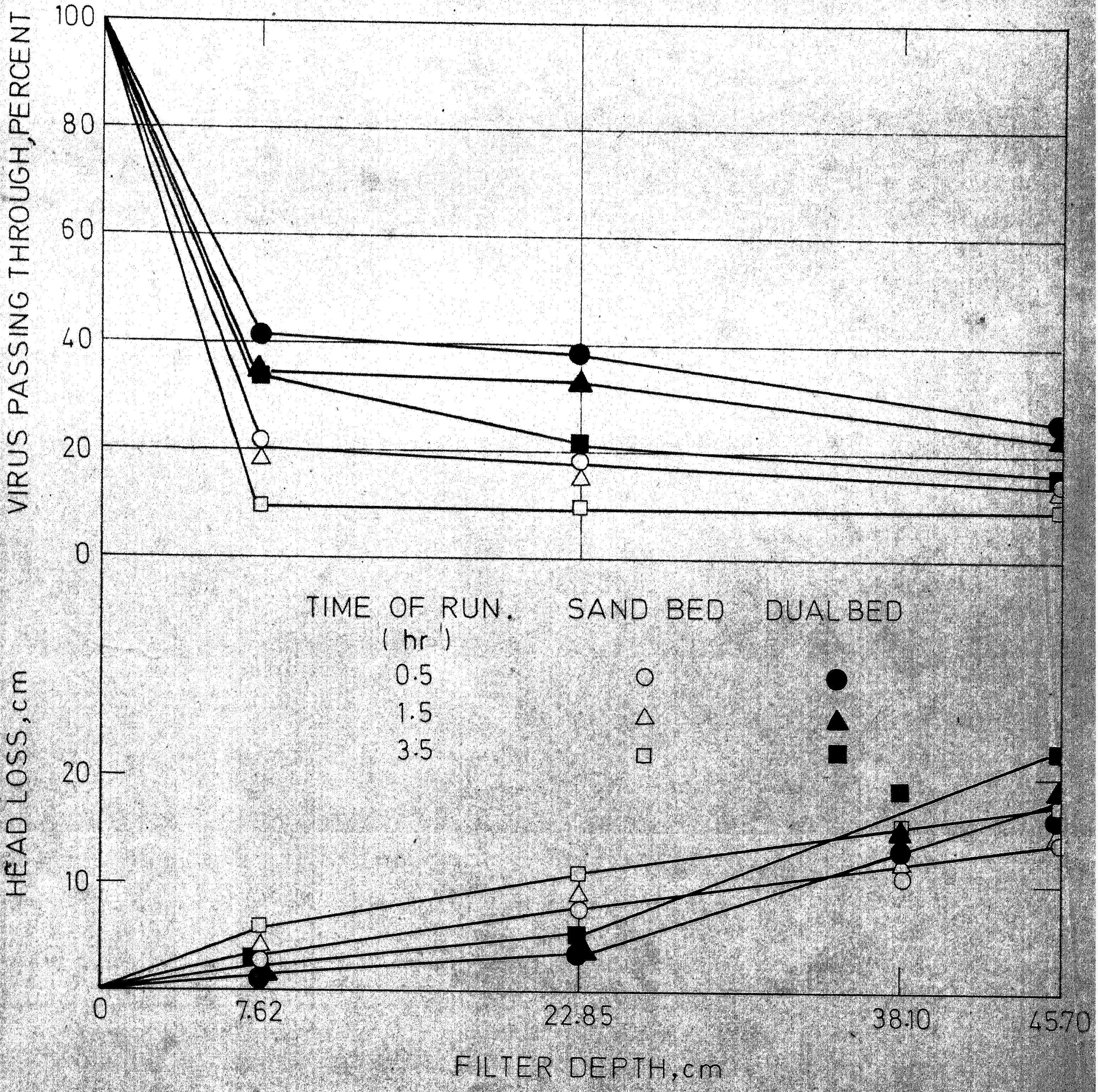

FIG. 24 FILTRATION OF VIRUS AND CLAY TURBIDITY THROUGH C-41 TREATED SAND AND DUAL MEDIA (Ref: Tables A30, A31)

TABLE 17

COMPARISON OF DISCRETE VIRUS FILTRATION WITH FILTRATION OF VIRUS IN ASSOCIATION WITH CLAY TURBIDITY

Media	λ_o		λ_o (Virus in Association With Clay) / λ_o (Discrete Virus)	Virus Removal, Percent (Clean Bed Condition)	
	Discrete Virus	Virus in Association With Clay		Discrete Virus	Virus in Association With Clay
Untreated Sand	0.006	0.030	5.1	22.75	73.00
C-41 Treated Sand	0.027	0.043	1.6	69.40	86.00

(Data Source: Tables A9, A17, A30).

(virus in association with clay turbidity) and the filter coefficient improved by nearly 5 times. This suggests that a major fraction of the virus particles are attached to the clay particles and are removed along with clay turbidity. Virus in association with clay turbidity may get attached (adsorbed) to the clay particles due to the formation of a clay-cation-virus bridge as hypothesized by Carlson et al. (1968). They observed 99, 90 and 98 percent virus attachment for 0.03 M NaCl, 0.005 M $CaCl_2$ and 5 x 10^{-6} M $Al_2(SO_4)_3$, respectively. The water used in this study (Table 11) contains significant amounts of monovalent (6.25 mM) and divalent (2 mM) cations and hence such virus attachment is possible. Therefore, the improvement in virus removal or filter coefficient is due to the improved transport of the bigger clay particles compared to the discrete virus particles. For filtration through C-41 treated sand, the percent virus removals were 69.4 for discrete virus and 86.0 for virus in association with clay turbidity and the filter coefficient improved 1.6 times. C-41 treatment of media changed its negative zeta potential to positive and thus improved the attachment step. While the percent removal of discrete virus improved from 22.75 to 69.4 following polyelectrolyte treatment of filter media the percent improvement in the removal of virus in association with clay was only from 73.0 to 86.0. This implies that all the discrete virus particles

that were transported did not get attached onto untreated media and improvement in attachment condition greatly improves discrete virus removal and probably similar submicron particles. The removal of 0.5 to 2 μm clay particles with attached viruses seem mostly due to mechanical straining in the interstices of top layers and hence improved attachment conditions do not show significant increase in virus removal.

Table 18 presents the results of filtration of virus and clay turbidity through single-media sand and dual-media coal-sand filters. The overall initial virus removals were 73.0, 86.0, and 97.5 and 67.0, 74.5, and 92.5 percent for filtration through sand (4.9 m/hr) and coal-sand dual-media (9.8 m/hr) filters, respectively under the three different filtration conditions. The corresponding turbidity removals were 97.8, 96.9, and 99.8 and 84.1, 77.3, and 99.2 percent, respectively. Percent turbidity removal was always more than the percent virus removal in both single and dual-media filtration. Also, it is seen that the filter coefficients calculated from turbidity removals were always more than those calculated using the virus removal data. These suggest that all the virus particles may not be attached to clay turbidity and a fraction may still remain as discrete virus particles.

TABLE 18

FILTRATION OF VIRUS AND CLAY TURBIDITY THROUGH SAND AND DUAL-MEDIA FILTERS

Media and Rate of Filtration	Filter Performance	Filtration Condition		
		Untreated Media	C-41 Treated Media	Untreated Media Following Coagulation and Settling
Sand 4.9 m/hr	λ_o (Virus)	0.03	0.043	0.035
	λ_o (Turbidity)	0.08	0.076	0.133
	Percent Virus Removal	73.00	86.000	97.500
	Percent Turbidity Removal	97.80	96.900	99.800
Coal-Sand 9.8 m/hr	λ_o (Virus)	0.028	0.030	0.057
	λ_o (Turbidity)	0.040	0.033	0.106
	Percent Virus Removal	67.000	74.500	92.500
	Percent Turbidity Removal	84.100	77.300	99.200

(Data Source: Tables A30, A31).

It is also evident from Table 18 that following coagulation and settling the performance (both virus and turbidity removal) of the sand filter at 4.9 m/hr is comparable to that of the dual-media coal-sand filter at 9.8 m/hr. This could be due to the better attachment potential of coal that formed the top half of the dual-media filter. A detailed comparison of dual-media coal-sand filter with single-media sand filter is presented in Table 19. It is evident that under the various filtration conditions the removal in the bottom half of the single media filter is quite negligible whereas in the dual-media filter the bottom half contributes significantly to the overall removal. Also, it is evident that eventhough the top half of the dual-media filter results in removals of a similar order as produced by the top half of the single-media filter, the headloss is 2 to 3 times less inspite of the higher rate of filtration.

Agrawal (1966) reported results of clay filtration (bentonite clay; d_p = 30 μm; ζ_p = -26 mV) through sand media at 4.9 m/hr filtration rate where he observed that the effluent quality deteriorated with time for both untreated as well as polyelectrolyte treated media. However, the filter performance improved with time when either 0.5 m$\underline{M}$ $CaCl_2$ or 10 mg/l alum was added just ahead of the filter to the suspension made in tap water which was very low in total dissolved solids (60 to 65 mg/l) and conductance (55 to

TABLE 19

COMPARISON OF SAND AND DUAL-MEDIA FILTERS FOR FILTRATION OF VIRUS IN ASSOCIATION WITH CLAY TURBIDITY

Description of Filter Operation	Virus Removal, Percent		Headloss, cm	
	Top Half	Bottom Half	Top Half	Bottom Half
(pH ≈ 8; Λ ≈ 900 μmhos/cm; Temp. ≈ 30°C; d_p = 0.5 to 2 μm; and ζ_p = −19 mV)				
Untreated Sand 4.9 m/hr	72.80	0.2	6.60	6.4
Untreated Dual-Media 9.8 m/hr	56.40	15.4	2.60	12.9
C-41 Treated Sand 4.9 m/hr	81.48	4.4	7.50	6.6
C-41 Treated Dual-Media 9.8 m/hr	61.70	12.6	3.60	12.9
(pH ≈ 7.8; Λ ≈ 900 μmhos/cm; Temp. ≈ 30°C; d_p = 2 to 5 μm; and ζ_p = −16 mV)				
Untreated Sand Bed Following Coagulation and Settling 4.9 m/hr	75.20	-	9.20	9.3
Untreated Dual-Media Following Coagulation and Settling 9.8 m/hr	82.60	10.0	3.00	14.3

(Data Source: Table A30, A31).

59 μmhos/cm). The turbid influent water in the present study was prepared with the tube well water which contains 2 m<u>M</u> of divalent cations in addition to 6.25 m<u>M</u> of monovalent cations. As mentioned already the filter performance under all filtration conditions was either almost constant or showed an improvement with time and hence the results of this investigation are in agreement with those obtained by Agrawal.

To simulate field conditions filter runs of 24 hr or more duration were made using low influent virus densities (Table 20). The influent turbidity and virus varied over a range of 33 to 62 FTU, and 52 to 100 PFU/ml in sand filtration, and 44 to 107 FTU, and 32 to 210 PFU/ml in dual-media filtration, respectively. It is evident that the sand filter effected turbidity and virus removals of 98 to 99.9 percent and 93 to 99.7 percent, respectively. The removal by the dual-media filter was over a range of 92 to 98 percent in the case of turbidity and 91 to 98.5 percent in the case of virus. The dual-media filter operating at 9.8 m/hr resulted in a headloss of about 0.95 m in 30 hr whereas the sand filter operating at half the filtration rate recorded a headloss of about 1.45 m in 23 hr.

TABLE 20

FILTRATION OF TURBID WATER WITH LOW VIRUS DENSITY THROUGH SAND AND DUAL-MEDIA FOLLOWING COAGULATION AND SETTLING

Filtration Condition	Time (hr)	PFU/ml		Virus Passing Through, Percent	Turbidity (FTU)		Turbidity Passing Through Percent	Headloss (cm)			
		Influent	Effluent		Inf	Eff		0-7.62 cm	0-22.85 cm	0-38.10 cm	0-45.70 cm
	0	100.6			45.0						
	0.5	52.0	3.530	6.80	43.5	0.05	0.11	2.9	7.5	11.9	12.9
0.5 mm Sand	4.0	53.0	0.681	1.28	33.0	0.05	0.15	10.3	15.0	18.7	22.5
4.9 m/hr	7.0	56.0	2.510	4.48	62.0	0.35	0.56	17.2	22.2	26.8	29.7
	17.0	58.6	0.336	0.57	37.0	0.10	0.27	88.8	95.8	100.3	103.5
	23.0	96.0	0.280	0.29	62.0	1.25	2.02	128.0	138.0	142.5	145.0
	0				44.0						
	0.5	149.0	5.825	3.91	44.0	2.60	5.90	1.2	3.5	13.1	17.4
	2.0	168.6	5.375	3.19	49.0	3.00	6.12	1.3	4.2	14.9	19.3
Dual-Media	4.0	184.0	6.125	3.33	65.0	3.00	4.62	1.6	4.8	17.0	21.6
1.0 mm Coal	7.0	210.0	3.050	1.45	107.0	2.62	2.45	3.8	7.6	22.0	26.7
(22.85 cm)	17.0	31.4	2.940	9.36	48.0	2.65	5.52	13.3	21.8	44.7	50.0
0.5 mm Sand	24.0	50.0	2.560	5.12	47.0	3.85	8.20	22.4	35.4	61.6	67.2
(22.85 cm)	30.0	32.0	2.690	8.40	62.0	5.25	8.47	34.5	53.5	87.6	92.5
9.8 m/hr											

7. DOMINANT MECHANISMS OF VIRUS REMOVAL BY FILTRATION

Particle removal in filtration, as discussed before, is conceived to consist of two steps: (i) the transport step wherein the particles from the bulk flow are brought close to the surface of the filter media grains, and (ii) the attachment step which is the actual adsorption step. Formulation of filter efficiency equations should consist of terms representing both these steps.

When viruses are present in association with turbidity, a major fraction may be attached to the particles causing turbidity while the remaining fraction may exist as discrete particles. The discrete virus particles are probably transported by diffusion (Table 8) whereas the dominant transport mechanisms for the attached viruses are presumably mechanical straining in the interstices, settling and interception. The last three mechanisms which are effected by the size of the suspended particles result in decreasing particle removal along the depth of a filter. In addition, Agrawal (1966) showed that the electrokinetie phenomena is a significant removal mechanism in filtration. Of the attachment forces, the molecular (Van der Waal's) forces lead to attraction, though they are of extremely limited range (50 nm) and little can be done to influence them. The electrical double layers can lead to attraction

or repulsion depending on whether the particles and media grain surfaces have potentials of unlike or like sign. The range of this interaction in highly conductive waters, such as wastewater effluents, is reported to be of the order of 10 nm (Ives, 1970) and probably, in no case, will be more than 50 nm in natural waters. In addition, an uncoated filter bed (negative zeta potential) is much less efficient compared to when it is coated (positive zeta potential) under similar transport conditions. In the overall removal process, attachment plays an equally important role as transport and can be improved by appropriate methods.

The results and observations presented in Chapter 6 for filtration of discrete virus and virus in association with clay turbidity reveal that the removal of discrete virus particles is generally poor and spread over the entire depth of the filter whereas the removal of virus in association with clay turbidity is significant and mostly confined to the top layers of the filter. Also, it is evident that while the nature of the media surface is a significant parameter for discrete virus removal as seen from the highly increased efficiency for media treated with alum or polyelectrolytes the nature of the media surface is not that significant for filtration of virus in association with clay turbidity (Table 17). This indicates dominance of different mechanisms in the removal of discrete virus and the virus in association with particulate matter causing turbidity.

Equation 2 can be modified as follows:

$$-\frac{\partial C}{\partial L} = \frac{1.5(1-f_o)}{d_m}\cdot\eta\cdot\alpha\cdot C + \begin{array}{l}\text{removal due to}\\ \text{electrokinetic}\\ \text{phenomena}\end{array} + \begin{array}{l}\text{removal due to}\\ \text{mechanical}\\ \text{straining}\end{array} \qquad 29$$

where, η = overall single collector efficiency (transport) which consists of single collector efficiency due to diffusion (η_D) single collector efficiency due to interception (η_I) and single collector efficiency due to gravity (η_G). The removal by mechanical straining is purely a physical phenomena. For a given filter bed, it depends only on the sizes of the particles in suspension and is expected to be independent of the chemical characteristics of the suspension. Therefore, the removals by this mechanism are to be considered separately. Theoretically, the collision efficiency factor (α) can vary from 0 to 1.

Integration of the above equation leads to:

$$\operatorname{Ln}\frac{C}{C_o} = -1.5(1-f_o)\cdot\eta\cdot\alpha\left(\frac{L}{d_m}\right) + \begin{array}{l}\text{removal due to}\\ \text{electrokinetic}\\ \text{phenomena}\end{array} + \begin{array}{l}\text{removal due}\\ \text{to mechanical}\\ \text{straining}\end{array} \qquad 30$$

that is, $\lambda = \lambda_E + \lambda_T + \lambda_S$ 31

where, λ = overall filter coefficient

λ_E = filter coefficient due to electrokinetic phenomena

λ_T = filter coefficient due to the other transport mechanisms

λ_S = filter coefficient due to removal by mechanical straining.

The computed values of filter coefficient (λ_o) and geometric mean concentration (C) of discrete virus particles in different layers of a filter are shown in Table 21 and in Fig. 25 and 26. These figures show that λ_o does appear to bear a straight line relationship to C for both untreated and alum floc treated media thereby indicating significance of electrokinetic phenomena in discrete virus removal. Considering Fig. 25, the equations of the two straight lines are:

$$\lambda_o = 2.875 \times 10^{-6} C - 0.01 \quad \text{(for untreated sand)} \qquad 32$$

$$\lambda_o = 2.875 \times 10^{-6} + 0.0185 \quad \text{(for alum treated sand)} \qquad 33$$

It can be assumed that electrokinetic phenomena and diffusion are the overall dominant mechanisms leading to the removal of discrete virus particles and therefore:

TABLE 21

FILTER COEFFICIENTS (λ_o) AND MEAN CONCENTRATIONS OF DISCRETE VIRUS PARTICLES (C) IN DIFFERENT LAYERS OF A FILTER

Media	λ_o, cm^{-1}/C, PFU/ml x 10^3					
	0-6.35 cm (0-2.5 in.)	6.35-21.60 cm (2.5-8.5 in.)	21.60-44.40 cm (8.5-17.5 in.)	0-21.60 cm (0-8.5 in.)	0-44.40 cm (0-17.5 in.)	6.35-44.40 cm (2.5-17.5 in.)
Untreated Sand	$\frac{0.0074}{6.100}$	$\frac{0.0076}{5.620}$	$\frac{0.0042}{5.050}$	$\frac{0.0076}{5.750}$	$\frac{0.0058}{5.480}$	$\frac{0.0056}{5.340}$
Alum Floc Treated Sand	$\frac{0.0592}{4.880}$	$\frac{0.0280}{3.280}$	$\frac{0.0245}{2.100}$	$\frac{0.0369}{3.950}$	$\frac{0.0285}{3.125}$	$\frac{0.0235}{2.600}$
Untreated Coal	$\frac{0.0577}{5.610}$	$\frac{0.0278}{3.925}$	$\frac{0.0105}{2.810}$	$\frac{0.0353}{4.540}$	$\frac{0.0222}{4.025}$	$\frac{0.0174}{3.480}$
Alum Floc Treated Coal	$\frac{0.0767}{5.420}$	$\frac{0.0447}{3.020}$	$\frac{0.0405}{1.382}$	$\frac{0.0540}{3.855}$	$\frac{0.0458}{2.490}$	$\frac{0.0408}{1.960}$

A

(Data Source: Tables A9, A10, A12, A13)

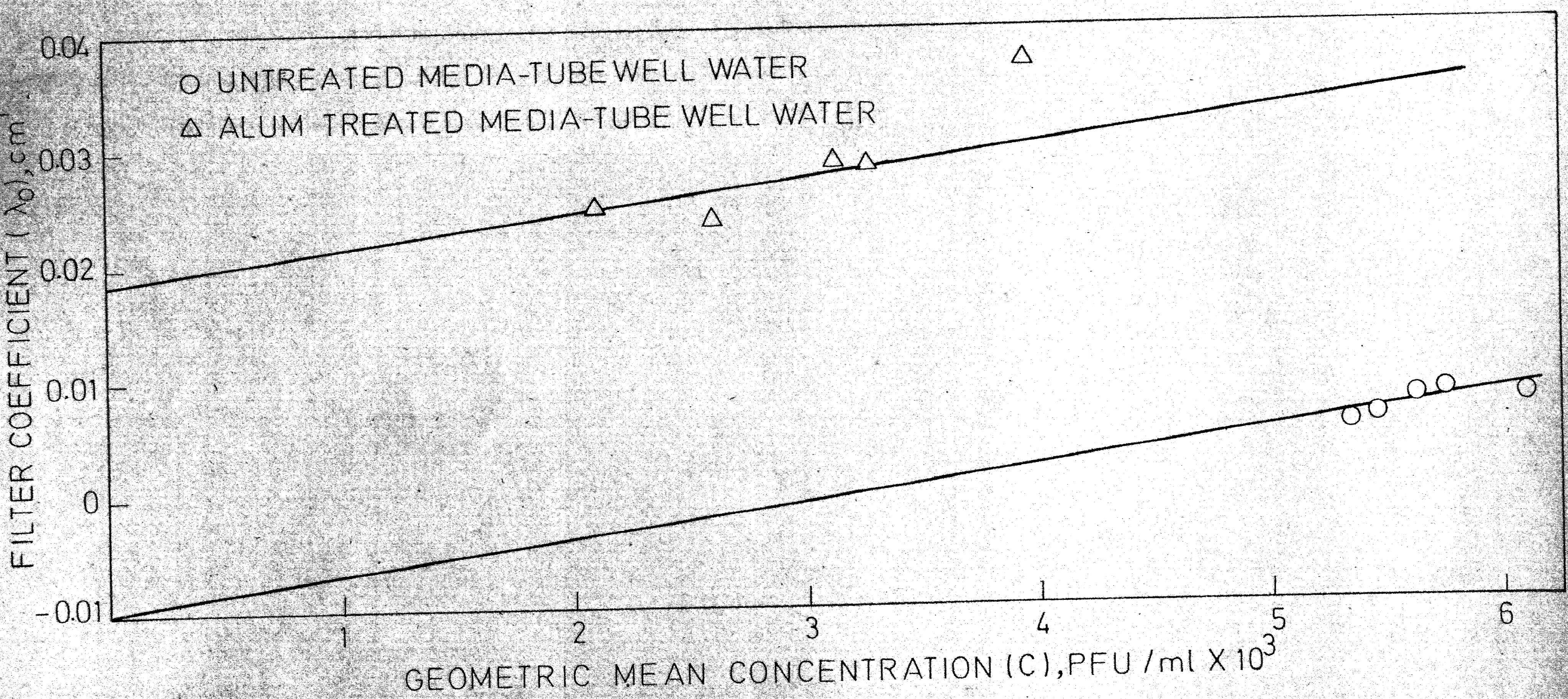

FIG.25 VARIATION OF FILTER COEFFICIENT (λ_0) WITH GEOMETRIC MEAN CONCENTRATION OF DISCRETE VIRUS PARTICLES IN A 0.5 mm SAND FILTER

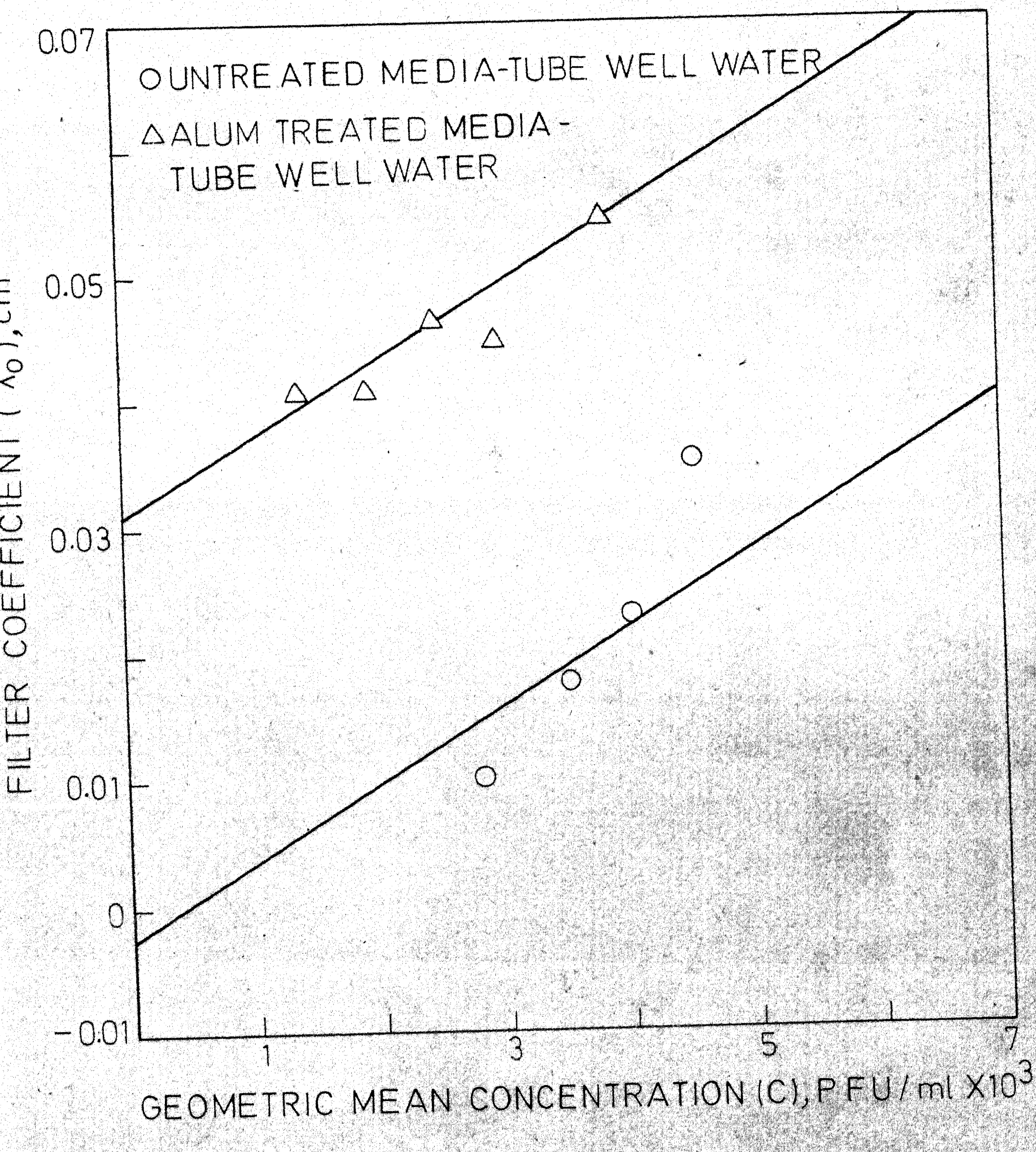

FIG. 26 VARIATION OF FILTER COEFFICIENT (λ_0) WITH GEOMETRIC MEAN CONCENTRATION OF DESCRETE VIRUS PARTICLES IN A 0.5mm COAL FILTER

$$\lambda_o = \lambda_{oE} + \lambda_{oD}$$

$$= \frac{7(1-f_o)}{d_m} \left\{ \frac{K_2 \ \zeta_p^2 \ C \ d_m}{\mu V_o} - \frac{K_1 \ \zeta_m \ \zeta_p \ d_p}{\mu V_o} \right\} + \lambda_{oD} \qquad 34$$

Substituting $f_o = 0.45$, $d_m = 0.05$ cm, $d_p = 25 \times 10^{-7}$ cm, $\mu = 10^{-2}$ poise and $V_o = 0.136$ cm/sec for the experimental sand filter, the above equation becomes:

$$\lambda_o = 2.83 \times 10^3 \ K_2 \ \zeta_p^2 \ C - 0.1415 \ K_1 \ \zeta_m \ \zeta_p + \lambda_{oD} \qquad 35$$

As mentioned earlier, the mass transport by the moving liquid predominates over the concentration dependent molecular diffusion and therefore it can be assumed that the removal due to diffusion is independent of the concentration of particles. Therefore, comparing the concentration dependent terms of Equations35 and 32:

$$2.83 \times 10^3 \ K_2 \ \zeta_p^2 = 2.875 \times 10^{-6}$$

and therefore,

$$K_2 = 3.14 \times 10^{-12} \quad (\text{for } \zeta_p = -18 \text{ mV})$$

In order to be able to estimate the value of K_1 in Equation 34, an assumption is required that the influence

of the zeta potential of filter media (ζ_m) on the diffusion contribution to the filter coefficient does not grossly affect such estimation. Further, a nominal error in the estimated value of K_1 will not significantly influence the computation of the electrokinetic contribution to the filter coefficient due to the fact that the term containg K_1 consists of d_p which is five orders of magnitude smaller than d_m contained by the first term. Therefore, from Equations 35, 32 and 33:

$$-0.01 = -0.1415\, K_1\, \zeta_p(-18.5) + \lambda_{oD}$$

$$+0.0185 = -0.1415\, K_1\, \zeta_p(+19.5) + \lambda_{oD}$$

Solving the above equations for a suspended particle zeta potential (ζ_p) value of -18 mV:

$$K_1 = 2.94 \times 10^{-4}$$

The contribution of electrokinetic phenomena in the removal of discrete virus particles would, therefore, be given by the expression:

$$\lambda_E = \frac{7(1-f_o)}{d_m}\left\{\frac{3.14 \times 10^{-12}\, \zeta_p^2\, C\, d_m}{\mu V_o} - \frac{2.94 \times 10^{-4}\, \zeta_p\, \zeta_m\, d_p}{\mu V_o}\right\} \qquad 36$$

in which ζ_p, ζ_m are in mV, C is geometric mean concentration of particles in PFU/ml and d_m, d_p, μ, V_o are in cgs units. The overall removal equation becomes:

$$\lambda_o = \frac{7(1-f_o)}{d_m} \{ \frac{3.14 \times 10^{-2} \zeta_p^2 C d_m}{\mu V_o} - \frac{2.94 \times 10^{-4} \zeta_p \zeta_m d_p}{\mu V_o} \} + \lambda_{oD} \qquad 37$$

A similar derivation for filtration through 0.5 mm coal filter (f_o = 0.5, ζ_m = -23.1 mV (untreated) and +38.1 mV (alum floc treated), and ζ_p = -18 mV) results in the following expression:

$$\lambda_o = \frac{7(1-f_o)}{d_m} \{ \frac{7.3 \times 10^{-12} \zeta_p^2 C d_m}{\mu V_o} - \frac{2.64 \times 10^{-4} \zeta_p \zeta_m d_p}{\mu V_o} \} + \lambda_{oD} \qquad 38$$

Using Equation 37 or 38, the contribution by diffusion (λ_{oD}) can be computed by subtracting the contribution by electrokinetic phenomena (λ_{oE}) from the experimental λ_o values. This value of λ_{oD} can be equated to the theoretical value of filter coefficient due to diffusion (λ_{oDT}) multiplied by the collision efficiency factor (α) and thereby the probable value of α can be estimated. The computed filter coefficient values due to electrokinetic mechanism and the probable

contribution by diffusion in discrete virus removal are presented in Table 22. For the sand filter, λ_{oE} increased 5 times (0.0105/0.0021) when the filter media was treated with alum. This improvement is expected as the second term due to electrokinetic contribution in Equation 37 becomes additive to the first term due to the positive media zeta potential (ζ_m) and negative particle zeta potential (ζ_p). Also, the filter coefficient due to diffusion increased 4.8 times (0.0180/0.0037). For the coal filter, the improvement in filter coefficient due to electrokinetic phenomena was 3.325 (0.0346/0.0104) times whereas there was no improvement in the filter coefficient due to diffusion following alum treatment. This indicates once again that coal has a superior attachment potential presumably due to more reactive surface groups compared to sand and a change in ζ_m may not influence the removal by diffusion. It is evident from Table 22 that the overall removal in discrete virus filtration can be satisfactorily accounted for by electrokinetic phenomena and diffusion.

Of the two mechanisms under consideration, diffusion is greatly improved with increasing temperature. It is seen from Table 22 that the contribution due to electrokinetic mechanism is insignificant whereas the contribution by diffusion accounts for the total virus removal observed in filtration through untreated sand at 36°C. At 36°C the

TABLE 22

CONTRIBUTION BY ELECTROKINETIC PHENOMENA AND DIFFUSION IN DISCRETE VIRUS FILTRATION THROUGH 0.5 mm SAND AND COAL

Media Condition and Zeta Potential	λ_o (Computed from Experimental Data	Filter Coefficient Due to		Theoretical φφφ Value of Filter Coefficient Due to Diffusion	Percent Collision Efficiency Factor, α
		Electrokinetic Phenomena φ	Diffusion φφ		
		λ_{oE}	λ_{oD}	λ_{oDT}	$\lambda_{oD}/\lambda_{oDT}$
Tube Well Water; pH 8; Temp. 30°C					
Untreated Sand $\zeta_m = -18.5$ mV	0.0058	0.0021	0.0037	0.0580	6.48
Alum Floc Treated Sand $\zeta_m = +19.5$ mV	0.0285	0.0105	0.0180	0.0580	31.05
Untreated Coal $\zeta_m = -23.1$ mV	0.0222	0.0104	0.0118	0.0580	22.4
Alum Floc Treated Coal $\zeta_m = +31.8$ mV	0.0458	0.0346	0.0112	0.0580	21.25
Tube Well Water; pH ≈ 8; Temp. ≈ 36°C					
Untreated Sand	0.02235	negligible	0.02235	0.0743	30.1

φ Computed from Equation 37 or 38

φφ $(\lambda_o - \lambda_{oE})$

φφφ *Computed from Equation in Table 8*

(Data Source: Tables A9, A10, A12, A13, A21).

collision efficiency factor improved to 30.1 percent due to probable ionization of more surface groups.

From the foregoing discussion it is evident that Equations 37 and 38 derived following the approach of Agrawal (1966) responded satisfactorily to discrete virus filtration through sand and coal. Diffusion and electrokinetic phenomena are the two dominant mechanisms for the removal of discrete virus and presumably similar submicron particles in filtration. However, diffusion becomes more predominant at higher temperatures.

In interpreting the data on the removal of viruses present in association with turbidity the removal of the particles causing turbidity is of primary concern since a major fraction of the viruses are attached to such particles. However, in such systems also it may be assumed that the virus concentration in PFU/ml indicates the number of the particles causing turbidity with attached viruses because only a minor fraction of the viruses may exist as discrete particles. Therefore, in the absence of adequate data on the concentration of the particles causing turbidity, the virus data may be used in the analysis. The contribution by diffusion in the transport step in such systems is negligible due to the larger size of the particles. The electrokinetic phenomena is also not significant as evidenced by the poor dependence of the filter coefficient

(λ_o) on the concentration of the particles (C) in both single-media sand and dual-media coal-sand filtration (Table 23). The remaining candidates which may dominate the overall mechanism of removal are mechanical straining, gravity settling and interception. Particle removal is normally confined to the top layers of the filter when these transport mechanisms are dominant and such trend was observed in this study (Fig. 22 and 24). However, for the observed particle size (0.5-5 μm) interception is insignificant and of mechanical straining and gravity settling the former is more significant (Table 8). It is to be noted that the ultimate removal of the particles transported by gravity settling depends significantly on the collision efficiency factor (α) which may vary between 0 and 1. The filter coefficients (λ_o) computed from the experimental turbidity data can be satisfactorily accounted for by mechanical straining and gravity settling with the dominance of the former mechanism (Table 24). However, for coagulated and settled water the significance of gravity settling may increase compared to when such pretreatment is not used.

In summary, mechanical straining and gravity settling are the dominant mechanisms leading to the removal of the particles causing turbidity and the viruses in association with such particles. However, diffusion and electrokinetic phenomena govern the removal of those virus particles which may exist as discrete particles.

TABLE 23

FILTER COEFFICIENTS (λ_o) AND MEAN CONCENTRATIONS OF PARTICLES (C) IN DIFFERENT LAYERS OF A FILTER

Media	λ_o, cm^{-1}/C, PFU/ml x 10^3		
	0-7.62 cm (0-3 in.)	7.62-22.85 cm (3-9 in.)	22.85-45.70 cm (9-18 in.)
Untreated Sand	$\frac{0.1650}{2.627}$	$\frac{0.0029}{1.370}$	$\frac{0.0003}{1.335}$
C-41 Treated Sand	$\frac{0.1990}{3.265}$	$\frac{0.0112}{1.405}$	$\frac{0.0121}{1.125}$
Untreated Dual-Media	$\frac{0.0097}{4.770}$	$\frac{0.0063}{3.150}$	$\frac{0.0192}{2.420}$
C-41 Treated Dual-Media	$\frac{0.1152}{3.890}$	$\frac{0.0054}{2.400}$	$\frac{0.0175}{1.890}$

(Data Source: Tables A30, A31).

TABLE 24

CONTRIBUTION BY MECHANICAL STRAINING AND GRAVITY SETTLING IN FILTRATION OF VIRUSES IN ASSOCIATION WITH CLAY TURBIDITY

Media	Observed Particle Size µm	λ_o Computed From Experimental Turbidity Data	Theoretical Value of Filter Coefficient Due to Mechanical Straining[φ] (λ_{oS})			Theoretical Value of Filter Coefficient Due to Gravity Settling[φφ] (λ_{oG})		
			0.5 µm	2 µm	5 µm	0.5 µm	2 µm	5 µm
Untreated Sand	0.5-2	0.080						
C-41 Treated Sand	0.5-2	0.076	0.0833	0.1362	0.5225	0.0020	0.0260	0.1600
Untreated Sand Following Coagulation and Settling	2-5	0.133						

φ Computed using Equation $\lambda_o = 26.5\ c\ d_p^{3/2}\ d_m^{-5/2}$ (Agrawal, 1966). Value of emprical factor c is assumed as 1.

φφ Table 8.

(Data Source: Table A30, A31).

8. SUMMARY AND CONCLUSIONS

The results of the present study show that single-media sand and dual-media coal-sand filters are effective in removing viruses in association with turbidity. However, for discrete virus particles (turbidity free water), both 0.5 mm sand and 1 mm coal showed poor virus removals (22.75 and 30.75 percent) at 4.9 m/hr. The removal efficiency doubled (61.5 percent) when 0.5 mm coal was used. Also, when the filter beds were treated with alum floc and cationic polyelectrolyte the virus removal efficiency of the sand filter increased to 71.8 and 69.4 percent, respectively and the corresponding values for the coal filter were 75.2 and 47.7 percent. Addition of polyvalent cations into the influent also showed improvement in virus removal with both untreated sand and coal. Furthermore, it was observed with untreated coal and sand that virus removal decreased at higher filtration rates and increased with an increase in temperature and decrease in pH value of the influent. Significant improvement in virus removal was observed during filtration of virus and clay turbidity. For untreated and cationic polyelectrolyte treated 0.5 mm sand filter (4.9 m/hr), virus removals were in the range 73.0 to 87.4 and 85.5 to 92.1 percent, respectively whereas turbidity removals in the range 82.5 to 97.4 and 91.9 to 96.9 percent were observed for

untreated and treated sand, respectively. The corresponding virus and turbidity removals were in the range 64.3 to 75.6 and 74.3 to 87.1 percent and 84.1 to 89.6 and 77.3 to 89.6 percent respectively for untreated and treated dual-media coal-sand filter (9.8 m/hr). A comparative study of single-media sand filter (0.5 mm) at 4.9 m/hr and dual-media coal-sand (1.0-0.5 mm) filter at 9.8 m/hr under simulated field condition with low influent virus density (52-210 PFU/ml) showed 93.2 to 99.7 and 90.6 to 98.5 percent virus removals and 98.0 to 99.9 and 91.5 to 97.5 percent turbidity removals during a filter run of 24 hr or longer.

Analysis of the virus filtration data indicated that the dominant mechanisms for removal of discrete virus particles are diffusion and electrokinetic phenomena. However, diffusion becomes more predominant at higher temperatures. The dominant mechanisms leading to the removal of the particles causing turbidity and the viruses in association with them (attached viruses) are mechanical straining and gravity settling. However, for coagulated and settled water, the significance of gravity settling may increase compared to when such pretreatment is not used.

Based on the findings of this investigation using a model virus, the following conclusions may be drawn:

(i) Single-media sand (4.9 m/hr) and dual-media coal-sand (9.8 m/hr) filters are effective in removing viruses

from turbid waters. Removals in the range 93.2 to 99.7 and 90.6 to 98.5 percent can be expected for sand and coal-sand filters, respectively following coagulation and flocculation.

(ii) Coal is more efficient than sand as filter media due to its greater attachment potential.

(iii) Virus removal decreases at higher filtration rate and increases with increase in temperature and decrease in pH value of the influent.

(iv) Viruses in association with turbidity (attached viruses) are more effectively removed than those which may exist as discrete particles.

(v) Significant virus removal can be expected by filtration through alum or cationic polyelectrolyte treated media without prior coagulation and settling. However, filtration following coagulation and flocculation is more efficient in removing viruses.

(vi) The dominant mechanismsfor removal of virus in association with turbidity are mechanical straining and gravity settling whereas the discrete viruses are removed by diffusion and electrokinetic phenomena.

(vii) Dual-media coal-sand filter may be recommended for effective virus and turbidity removal from the viewpoints of efficiency and economy.

9. ENGINEERING SIGNIFICANCE

The most significant result of this study is that a more complete understanding of the removal of viruses from water by filtration has been attained. This is very pertinent in view of our increasing concern regarding the occurrence and survival of viruses in raw, treated and renovated water supplies. The removal of viruses from water supplies becomes extremely important due to an everincreasing demand for water for public consumption and other uses which, in turn, will require more water reuse. A good understanding of the basic mechanisms involved in the removal of viruses from water by filtration and the role of various other parameters affecting the process should aid in developing design standards for water treatment facilities on a sound, realistic and rational basis.

The findings of this study that bituminous coal is superior to sand in virus sorption and that a dual-media coal-sand filter at 9.8 m/hr is comparable to a single-media sand filter at 4.9 m/hr in virus removal are quite significant for countries like India. In India, due to a lack of suitable anthracite deposits, use of bituminous coal in dual-media filtration has been suggested on the basis of turbidity and bacterial removal. The observation that a polyelectrolyte treated filter at a lower temperature is

comparable in virus removal efficiency to an untreated filter at a higher temperature suggests that such treatment may be desirable at lower temperatures especially during times of anticipated virus contamination of raw water supplies.

From the experimental results it is possible to extrapolate some generalizations which are of practical significance. The foremost among these is the interpretation of MS2 removal data in terms of viruses which may be more significant in water supplies, viz., human enteric viruses. In view of the observations of the Metropolitan Water Board, London (1971-73) regarding the parallel behaviour of poliovirus and MS2 phage during slow sand filtration, the data on virus removal by filtration obtained in the present study may be reasonably extrapolated in terms of removal of enteroviruses from water by filtration. Also, the dominant mechanisms for removal of discrete virus particles by filtration, viz., diffusion and electrokinetic phenomena may apply reasonably well to other submicron particles. The findings that viruses in association with turbidity are removed better in filtration and removal of such viruses is governed by the same mechanisms which control removal of turbidity suggests that in a filtration plant care should be taken to produce a high quality effluent in terms of turbidity. It further supports the recommendation of the Committee on Environmental Quality Management (1970) that

a turbidity of less than 0.1 JTU would provide a greater assurance of the absence of virus.

Finally, the results of this study suggests that rapid sand filtration as it is practiced today is effective in removing pathogenic viruses from water if proper care is taken to control various parameters which affect the process. However, it should be noted that a dual-media coal-sand filter is more desirable in terms of water quality and economy as well.

10. SUGGESTIONS FOR FURTHER WORK

On the basis of the results of the current investigation it is felt that further work should be pursued in the following areas.

(i) Studies should be conducted on a pilot scale to evaluate the virus removal efficiencies of single-media sand as well as dual-media coal-sand filters using a model virus, preferably an enterovirus. The influent virus concentrations in these studies should be comparable to that in natural waters which will require the use of suitable virus concentration techniques.

(ii) Existing sand filters should be evaluated in terms of their efficiencies in removing naturally occurring viruses using recently developed sophisticated virus concentrators. These studies should be conducted over a period of several years to evaluate the effects of temperature and other water quality parameters on virus removal by filtration.

(iii) Fate of viruses in the filter backwash water and possible hazards associated with its disposal should be studied.

(iv) Plant scale studies should be undertaken on dual-media coal-sand filter for assessing its reliability in performance in comparison with a single-media sand filter. Investigation on the hydraulics of backwashing of coal-sand filter and other necessary modification required for possible conversion of existing sand filters to coal-sand filters should be undertaken.

LIST OF REFERENCES

Adams, M.H. (1959). Bacteriophages. Interscience Publ., New York, N.Y.

Agrawal, G.D. (1966). Electrokinetic Phenomena in Water Filtration. Ph.D. Thesis, University of California, Berkeley, California.

American Water Works Association Handbook of Public Water Supplies (1971). Water Quality and Treatment. McGraw-Hill Book Co., New York, N.Y.

Arceivala, S.J., Lakshminarayana, J.S.S., Algarswamy, S.R., and Sastry, C.A. (1971). Waste Stabilization Ponds in India. Central Public Health Engineering Research Institute, Nagpur, India.

Baumann, E.R. and Oulman, C.S. (1964). Streaming Potentials in Diatomite Filtration of Water. J. Am. Water Wks. Assn., 56, 7, p. 915.

Baumann, E.R. and Oulman, C.S. (1964a). Progress Report, Project 388-S, Iowa Eng. Expt. Station, Ames, Iowa.

Baumann, E.R. and Oulman, C.S. (1969). Sand and Diatomite Filtration Practices. Preprint Project 735S, Eng. Res. Inst., Iowa State University, Ames, Iowa.

Baumann, E.R. and Oulman, C.S. (1970). Polyelectrolyte Coatings for Filter Media. Filtration and Separation, Proc. of the Filtration Society, p. 682.

Berg, G. (1966). Virus Transmission by the Water Vehicle. I. Viruses. Health Lab Sci., 3, p. 86.

Berg, G. (1966a). Virus Transmission by the Water Vehicle. II. Virus Removal by Sewage Treatment Procedures. Health Lab Sci., 3, p. 90.

Berg, G. (Ed.). (1966b). Transmission of Viruses by the Water Route. Interscience Publ., New York, N.Y.

Bierman, A. (1955). Electrostatic Forces between Nonidentical Colloidal Particles. J. Colloid Sci., 10, p. 231.

Boehm, H.P. (1966). Chemical Identification of Surface Groups. In Advances in Catalysis, Vol. 16, D.D. Eley, H.Pines, and P.B. Weisz,(Eds.), Academic Press, New York, N.Y.

Boyd, R.H. and Ghosh, M.M. (1974). An Investigation of the Influences of Some Physicochemical Variables on Porous Media Filtration. J. Am. Water Wks. Assn., 66, 2, p. 98.

Brown, J.K. (1955). The Infrared Spectra of Coals. J. Chem. Soc., p. 744.

Brown, T.S., Malina, J.F., and Moore, B.D. (1974). Virus Removal by Diatomaceous Earth Filtration - Part I. J. Am. Water Wks. Assn., 66, 2, p. 98.

Brunner, D.R. and Sproul, O.J. (1970). Virus Inactivation During Phosphate Precipitation. J. San. Eng. Div., Proc. Am. Soc. Civil Engr., 96, SA2, p. 365.

Burvas, S., Gaines, A.F., Husadsri, T., Prasertwayekij, A., and Sucharetakul, N. (1970). The Infrared Spectra of Tertiary Lignites. Fuel, 49, p. 180.

Bush, A.F. and Isherwood, J.D. (1966). Virus Removal in Sewage Treatment. J. San. Eng. Div., Proc. Am. Soc. Civil Engr., 92, SA1, p. 99.

Camp, T.R. (1964). Theory of Water Filtration. J. San. Eng. Div., Proc. Am. Soc. Civil Engrs., 90, SA1, p. 1.

Cannon, C.G. (1953). Infrared Spectra Coals and Coal Products. Nature, 171, p. 308.

Carlson, H.J., Ridenour, G.M., and McKhann, C.F. (1942). Efficacy of Standard Purification Methods in Removing Poliomyelitis Virus from Water. Am. J. Pub. Health, 32, p. 1256.

Carlson, G.F., Jr., Woodard, F.E., Wentworth, D.F., and Sproul, O.J. (1968). Virus Inactivation on Clay Particles in Natural Waters. J. Water Poll. Cont. Fed., 40, 2, Part 2, p. R89.

Carlton, S.S., Marr, C.M., Foye, C.W., Owens, C.K., Chiancone, R.A., and Raynes, B.C. (1965). Investigations on the Use of Coal for Treatment of Sewage and Wastewaters. Rand Development Corp., Cleveland, Ohio.

Carmon, P.C. (1940). Constitution of Colloidal Silica. Trans. Faraday Soc., 36, p. 964.

Chang, S.L., Stevenson, R.E., Bryant, A.R., Woodward, R.L., and Kabler, P.W. (1958). Removal of Coxsackie and Bacterial Viruses in Water of Known Chemical Content by Flocculation with Aluminium Sulphate or Ferric Chloride Under Various Testing Conditions. Am. J. Pub. Health, 48, p. 51 and 159.

Chang, S.L. (1968). Waterborne Viral Infections and Their Prevention. Bull. World Health Organization, 38, p. 401.

Chaudhuri, M. (1969). Virus Removal by Chemical Coagulation. Ph.D. Thesis, University of Illinois, Urbana, Ill.

Chaudhuri, M. and Engelbrecht, R. S. (1970). Virus Removal in Wastewater Renovation by Chemical Coagulation and Flocculation. In Advances in Water Pollution Research, Vol. II, S.H. Jenkins, (Ed.), Pergamon Press, Oxford.

Chaudhuri, M. and Engelbrecht, R.S. (1970). Removal of Viruses from Water by Chemical Coagulation and Flocculation. J. Am. Water Wks. Assn., 62, p. 563.

Chaudhuri, M. (1973). Virus Removal in Waste Stabilisation Ponds. Indian J. Env. Health., 16, 3, p. 171.

Chaudhuri, M. and Engelbrecht, R.S. (1974). Viruses in Water Supplies. J. Indian Water Wks. Assn., 6, 2, p. 87.

Chaudhuri, M., Amirhor, P., and Engelbrecht, R.S. (1974). Virus Removal by Diatomaceous Earth Filtration. J. Env. Eng. Div., Proc. Am. Soc. Civil Engrs., 100, EE4, p. 937.

Clarke, N.A. and Chang, S.L. (1959). Enteric Viruses in Water. J. Am. Water Wks. Assn., 51, 10, p. 1299.

Clarke, N.A., Stevenson, R.E., Chang, S.L., and Kabler, P.W. (1961). Removal of Enteric Viruses from Sewage by Activated Sludge Treatment. Am. J. Pub. Health, 51, 8, p. 1118.

Clarke, N.A., Berg, G., Kabler, P.W., and Chang, S.L. (1964). Human Enteric Viruses in Water: Source, Survival and Removability. In Advances in Water Pollution Research, Vol. II, W.W. Eckenfelder, Jr. (Ed.), Macmillan Co., New York, N.Y.

Clarke, N.A., Akin, E.W., Hill, W.F., Jr., Liu, O.C., Brashear, D.A., Hoff, J.C., and Jakubowski, W. (1974). Virus Study for Drinking Water Supplies. Presented at the 94th American Water Works Association Annual Conference, Boston, Mass.

Cleasby, J.L. and Baumann, E.R. (1962). Selection of Sand Filtration Rates. J. Am. Water Wks. Assn., 54, p. 579.

Cleasby, J.L. and Woods, C.F. (1974). Intermixing of Dual- and Multi-Media Granular Filters. Preprint Project 1048-S, Eng. Res. Inst., Iowa State University, Ames, Iowa.

Cliver, D.O. (1966). Enterovirus Detection by Membrane Chromotography. In Transmission of Viruses by the Water Route, G. Berg, (Ed.), Interscience Publ., New York, N.Y.

Cliver, D.O. and Herrmann (1972). Proteolytic and Microbial Inactivation of Enteroviruses. Water Res., 6, p. 797.

Coin, L., Menetrier, M.L., Labonde, J., and Hannoun, M.C. (1965). Modern Microbiological and Virological Aspects of Water Pollution. In Advances in Water Pollution Research, Vol. I, O. Jaag, (Ed.), Pergamon Press, New York, N.Y.

Committee Report (1970). Committee on Environmental Quality Management of the Sanitary Engineering Division. Engineering Evaluation of Virus Hazard in Water. J. San. Eng. Div., Proc. Am. Soc. Civil Engrs., 96, SA1, p. 111.

Conrat, H.F. (1968). The Small RNA Viruses of Plants, Animals and Bacteria - Chemical Properties. In Molecular Basis of Virology, H.F. Conrat, (Ed.), Reinhold Book Co., New York, N.Y.

Cookson, J.T., Jr. (1967). Adsorption of Viruses on Activated Carbon: Equilibria and Kinetics of the Attachment of E. coli Bacteriophage T4 on Activated Carbon. Env. Sci. and Tech., 1, 1, p. 1.

Cookson, J.T., Jr. (1969). Mechanism of Virus Adsorption on Activated Carbon. J. Am. Water Wks. Assn., 61, p. 52.

Cookson, J.T., Jr. (1970). Removal of Submicron Particles in Packed Beds. Env. Sci. and Tech., 4, 2, p. 128.

Drewry, W.A. and Eliassen, R. (1968). Virus Movement in Ground Water. J. Water Poll. Cont. Fed., 40, 2, p.R257.

Einstein, A. (1907-1909). Annals of Phys. Lpz., 17, p. 549; 19, p. 371; 22, p. 569.

Fair, G.M. and Geyer, J.C. (1954). Water Supply and Wastewater Disposal. John Wiley and Sons, New York, N.Y.

Filmer, R.W. and Corey, A.T. (1966). Transport and Retention of Virus Sized Particles in Porous Media. Sanitary Engineering Papers, Colorado State University, Fort Collins, Colorado.

Fitzpatrick, J.A. and Spielman, L.A. (1973). Filtration of Aqueous Latex Suspensions Through Beds of Glass Spheres. J. Coll. Interface Sci., 43, p. 2.

Friedel, R.A. and Pelipetz, M.G. (1953). Infrared Spectra of Coal and Carbohydrate Chars. J. Opt. Soc. Am., 43, p. 1051.

Friedel, R.A. and Queiser, J.A. (1956). Infrared Analysis of Bituminous Coals and Other Carbonaceous Materials. Anal. Chem., 28, p. 22.

Gelfand (1961). The Occurrence in Nature of the Coxsackie and ECHO Viruses. Progr. Med. Virol., 3, p. 193.

Gilcreas, F.W. and Kelly, S.M. (1955). Relation of Coliform Organism Test to Enteric Virus Pollution. J. Am. Water Wks. Assn., 47, p. 683.

Gillespie, T. (1955). On the Adhesion of Drops and Particles on Impact at Solid Surfaces - I. J. Colloid Sci., 10, p. 266.

Goodheart, C.R. (1969). An Introduction to Virology. W.B. Saunders Co., Philadelphia.

Grabow, W.O.K. (1968). The Virology of Wastewater Treatment. Water Res., 2, p. 675.

Gregory, J. (1964). Ph.D. Thesis, University of London, London.

Grinstein, S., Melnick, J.L., and Wallis, C. (1970). Virus Isolations from Sewage and from a Stream Receiving Effluents of Sewage Treatment Plants. Bull. World Health Organization, 42, p. 291.

Hall, W.A. (1957). Analysis of Sand Filtration. J. San. Eng. Div., Proc. Am. Soc. Civil Engrs., 83, SA3, p. 1276.

Harrison, L.B. (1931). Activated Carbon at Bay Cities Filtration Plant. J. Am. Water Wks. Assn., 23, p. 1388.

Hazen, A. (1904). On Sedimentation. Trans. Am. Soc. Civil Engr., 53, p. 63.

Heertjes, P.M. and Lerk, C.F. (1967). The Functioning of Deep Bed Filters. Trans. Inst. of Chem. Engrs., 45, p. 129.

Hill, W.F., Akin, E.W., and Benton, W.A. (1971). Detection of Viruses in Water: A Review of Methods and Application. Water Res., 5, p. 967.

Hill, W.F., Jr., Akin, E.W., Benten, W.H., Mayhew, C.J. and Jakubowski, W. Apparatus for Conditioning Unlimited Quantities of Finished Waters for Enteric Virus Detection. App. Microbiol., p. 1177.

Ison, C.R. (1967). Dilute Suspensions in Filtration. Ph.D. Thesis, University of London, London.

Ison, C.R. and Ives, K.J. (1969). Removal Mechanisms in Deep Bed Filtration. Chem. Eng. Sci., 24, p. 717.

Ives, K.J. (1960). Rational Design of Filters. Proc. Inst. Civil Engrs., 16, p. 189.

Ives, K.J. and Gregory, J. (1967). Basic Concepts of Filtration. Proc. Soc. Water Treat. Exam., 16, p. 147.

Ives, K.J. (1970). Advances in Deep Bed Filtration. Trans. Inst. Chem. Engrs., 48, p. 794.

Ives, K.J. (1971). Filtration of Water and Wastewater. CRC Critical Reviews in Environmental Control.

Iwasaki, T. (1937). Some Notes on Sand Filtration. J. Am. Water Wks. Assn., 29, p. 1591.

Johnson, G.E., Kunka, L.M., Forney, A.J., and Field, I.H. (1965). The Use of Coals and Modified Coals as Adsorbents for Removing Organic Contaminants from Wastewaters. Report No. 6884. U.S. Dept. Interior, Bureau of Mines.

Jorden, R.M. (1963). Electrophoretic Studies of Filtration. J. Am. Water Wks. Assn., 55, p. 771.

Kasel, J.A., Rosen, L., and Evans, H.E. (1963). Infection of Human Volunteers with a Reovirus of Bovine Origin as Quoted by Committee Report (1970).

Kelly, S. and Sanderson, W.W. (1959). The Effect of Sewage Treatment on Viruses. Sewage and Ind. Wastes, 31, p. 683.

Kelly, S., Sanderson, W.W., and Neidl, C. (1961). Removal of Entero Viruses from Sewage by Activated Sludge. J. Water Poll. Cont. Fed., 33, p. 1056.

Kempf, J.E., Wilson, M.G., Pierce, M.E., and Soule, M.H. (1942). Effect of Aluminium Hydroxide Sedimentation, Sand Filtration and Chlorination on the Virus of Poliomyelitis. Am. J. Pub. Health, 32, p. 1366.

King, P.H., McNeice, F.E., and Warren, P.S. (1969). Removal of Selected Pesticides from Water by Sorption on Coal. Bulletin 32, Water Resources Research Center, Virginia Polytechnique Institute, Blacksburg, Virginia.

Kraemer, H.F. and Johnstone, H.F. (1955). J. Ind. and Eng. Chem., 47, p. 2426.

Levich, V.G. (1962). Physicochemical Hydrodynamics. Prentice-Hall, Englewood Cliffs, N.J.

Levintow, J.W. and Darnell, J.E. (1960). A Simplified Procedure for Purification of Large Amounts of Poliovirus; Characterization and Aminoacid Analysis of Type 1 Poliovirus. J. Biol. Chem., 235, p. 70.

Mack, W.N., Frey, J.R., Riegle, B.J., and Mallmann, W.L. (1962). Entero Virus Removal by Activated Sludge Treatment. J. Water Poll. Cont. Fed., 34, p. 1133.

Mackrle, V. and Mackrle, S. (1961). Adhesion in Filters. J. San. Eng. Div., Proc. Am. Soc. Civil Engr., 87, SA3, p. 17.

Mackrle, V., Dracka, O., and Svec, J. (1965). Hydrodynamics of the Disposal of Low Level Liquid Radioactive Wastes in Soil. Internat. At. Energy Agency, Contract Report 98.

Maroudas, A. and Eisenklam, P. (1965). Clarification of Suspensions: A Study of Particle Deposition in Granular Media. Chem. Eng. Sci., 20, p. 867.

Mattson, J.S., Mark, H.B., Jr., Malbin, M.D., Weber, W.J., Jr., and Crittenden, J.C. (1969). Surface Chemistry of Active Carbon: Specific Adsorption of Phenols. J. Coll. Interface Sci., 31, p. 116.

Merrel, J.C., Jr. and Katko, A. (1966). Reclaimed Wastewater for Santee Recreational Lakes. J. Water Poll. Cont. Fed., 38, p. 1310.

Metropolitan Water Board, London (1971-73). Fortyfifth Report of the Director of Water Examination.

Mints, D.M. (1969). Preliminary Treatment of Water Before Filtration. Special Subject No. 6. I.W.S.A. Congress, Vienna.

Mosley, J.W. (1966). Transmission of Virus Diseases by Drinking Water. In Transmission of Viruses by the Water Route, G. Berg, (Ed.), Interscience Publ., New York, N.Y.

Neefe, J.R., Baty, J.B., Reinhold, J.G., and Stokes, J., Jr. (1947). Inactivation of the Virus of Infectious Hepatitis in Drinking Water. Am. J. Pub. Health., 37, p. 365.

Nupen, E.M. (1970). Virus Studies on the Windhock Wastewater Reclamation Plant. Water Res., 4, p. 661.

O'Melia, C.R. and Crapps, D.K. (1964). Some Chemical Aspects of Rapid Sand Filtration. J. Am. Water Wks. Assn., 56, p. 1326.

O'Melia, C.R. and Stumm, W. (1967). Theory of Water Filtration. J. Am. Water Wks. Assn., 59, p. 1393.

Overbeek, J.T.G. (1952). Experiments on Long Range Attractive Forces Between Macroscopic Objects. J. Coll. Sci., 1, p. 343.

Overby, L.R., Barlow, G.H., Doi, R.H., Jacob, M., and Spiegelman, S. (1966). Comparison of Two Serologically Distinct Ribonucleic Acid Bacteriophages. I. Properties of the Viral Particles. J. Bacteriol., 91, p. 442.

Oza, P.P. (1974). Removal of Viruses from Water by Sorption onto Coal. Ph.D. Thesis, Indian Institute of Technology, Kanpur.

Paramasivam, R., Gadkari, S.K., Joshi, N.S., and Deshpande, A.W. (1973). Bituminous Coal - A Substitute for Anthracite in Two Layer Filtration of Water. Indian J. Env. Health, 15, 3, p. 178.

Pirnie, M. (1931). Experience with Powdered Activated Carbon. J. Am. Water Wks. Assn., 23, 9, p. 1393.

Plotkin, S.A. and Katz, M. (1966). Minimal Infective Doses of Viruses for Man by the Oral Route. In Transmission of Viruses by the Water Route, G. Berg, (Ed.), Interscience Publ., New York, N.Y.

Puri, B.R. (1970). Surface Complexes on Carbons. In Chemistry and Physics of Carbon, P.L. Walker, Jr., (Ed.), Marcel Dekker, New York, N.Y.

Ranz, W.E. (1951). Tech. Report No. 3, Contract A-T-(30-3)-28 of University of Illinois.

Ranz, W.E. and Wong, J.B. (1952). Impaction of Dust and Smoke Particles on Surface and Body Collectors. J. Ind. and Eng. Chem., 44, 6, p. 1371.

Rao, N.U. (1971). Personal Communication.

Rao, V.C., Sullivan, R., Read, R.B., and Clarke, N.A. (1968). A Simple Method for Concentrating and Detecting Viruses in Water. J. Am. Water Wks. Assn., 60, p. 1288.

Rhead, T.F.E. and Wheeler, R.V. (1912). J. Chem. Soc., 101, p. 846.

Rhead, T.F.E. and Wheeler, R.V. (1913). J. Chem. Soc., 102, p. 461.

Robeck, G.G., Clarke, N.A., and Dostal, K.A. (1962). Effectiveness of Water Treatment Processes in Virus Removal. J. Am. Water. Wks. Assn., 54, p. 1275.

Rosenkranz, F.D. and Silveston, P.L. (1970). Use of Coal for Treatment of Pulping Waste. Can. Min. Met. Bull., p. 1.

Rubenstein, S.H., Orbach, H.G., Shuber, N., King, E., and Jackler, J. (1971). Freeze Concentration of Viral Agents from Large Volumes of Water. J. Am. Water Wks. Assn., 63, 5, p. 301.

Schafer, H.N.S. (1970). Carboxyl Groups and Ion Exchange in Low Rank Coals. Fuel, 49, p. 197.

Shannon, E. and Silveston, P.L. (1968). Studies on the Use of Coals for Waste Treatment. Chem. Eng. Progr. Symp. Ser., 64, p. 198.

Shekhtman, Y.M. (1961). Filtration of Suspensions of Low Concentration. Publishing House of the U.S.S.R. Academy of Sci., Moscow.

Shih, J.W., Osawa, Y., and Fujii (1972). Infrared Spectra of Heat Treated Coal. Fuel, 5, p. 153.

Simon, I. and McMahon, H.O. (1953). Study of the Structure of Quartz, Cristobalite, and Vitreous Silica by Reflection in Infrared. J. Chem. Phys., 21, p. 1.

Spalding, G.R. (1930). Activated Char as a Deodorant in Water Purification. J. Am. Water Wks. Assn., 22, 5, p. 646.

Spielman, L.A. and Friedlander, S.K. (1974). Role of the Electrical Double Layer in Particle Deposition by Convective Diffusion. J. Coll. Interface Sci., 46, 1, p. 22.

Sproul, O.J., Larochell, L.R., Wentworth, D.F., and Thorup, R.T. (1967). Virus Removal in Water Reuse Treating Processes. Chem. Eng. Progr. Symp. Ser., 63, 78, p. 130.

Sproul, O.J. (1971). Virus Inactivation by Water Treatment. J. Am. Water Wks. Assn., 64, 1, p. 31.

Sproul, O.J. (1973). Quality of Recycled Water: Fate of Infectious Agents. Can. Inst. Food Sci. Tech. J., 6, 2.

Standard Methods for the Examination of Water and Wastewater, (1971). 13th Edn., American Public Health Association, New York, N.Y.

Stanley, D.R. (1955). Sand Filtration Studies with Radio Tracers. J. San. Eng. Div., Proc. Am. Soc. Civil Engr., 81, p. 592.

Stein, P.C. (1940). A Study of the Theory of Rapid Filtration of Water Through Sand. Ph.D. Thesis, M.I.T., Cambridge, Mass.

Stern (1931). Water Works and Sewerage, 78, p. 11.

Stumm, W. and Morgan, J.J. (1962). Chemical Aspects of Coagulation. J. Am. Water Wks. Assn., 54, p. 971.

Sweet, B.H., McHale, J.S., Hardy, K.J., and Klein, E. (1971). Concentration of Virus from Water by Electro-Osmosis and Forced Flow Electrophoresis. Biochem., p. 77.

Sweet, B.H., McHale, J.S., Hardy, K.J., Morton, F., Smith, J.K., and Klein, E. (1971a). Concentration of Virus from Water by Osmotic Ultrafiltration.- I. Water Res., 5, p. 283.

Technical Digest. (1971). Bituminous Coal - A Substitute Filter Media for Anthracite, Central Public Health Engineering Research Institute, Nagpur, India.

Thorup, R.T., Nixon, F.P., Wentworth, D.F., and Sproul, O.J. (1970). Virus Removal by Coagulation with Polyelectrolytes. J. Am. Water Wks. Assn., 62, 2, p. 97.

Trowbridge (1931). Pub. Wks., 62, 6, p. 19.

Van Vucht, H.A., Rietveld, B.J., and Vankrevelon, D.W. (1955). Chemical Structure and Properties of Coal VIII - Infrared Absorption Spectra. Fuel, 34, p. 50.

Wallis, C., Melnick, J.L., and Fields, J.E. (1970). Detection of Viruses in Large Volumes of Natural Waters by Concentration on Insoluble Polyelectrolytes. Water Res., 4, p. 787.

Wallis, C., Melnick, J.L., and Fields, J.E. (1971). Concentration and Purification of Viruses by Adsorption to and Elution from Insoluble Polyelectrolytes. Appl. Microbiol., 21, p. 703.

Wallis, C., Homma, A., and Melnick, J.L. (1972). A Portable Virus Concentrator for Testing Water in the Field. Water Res., 6, p. 1249.

Weber, W.J., Jr.,(Ed.). (1972). Physicochemical Processes for Water Quality Control. Wiley-Interscience, New York, N.Y.

Willard, H.H., Merritt, L.L., Jr., and Dean, J.A. (1965). Instrumental Methods of Analysis. D. Van Nostrand Co. Inc., New York, N.Y.

Yao, K.M. (1968). Influence of Suspended Particle Size on the Transport Aspect of Water Filtration. Ph.D. Thesis, University of North Carolina, Chappel Hill, N.C.

Yao, K.M., Habibian, M.T., and O'Melia, C.R. (1971). Water and Wastewater Filtration - Concepts and Applications. Env. Sci. Tech., 5, 11, p. 1105.

Young, R.H.F. and Burbank, N.C., Jr. (1973). Virus Removal in Hawaiian Soils. J. Am. Water Wks. Assn., 65, p. 598.

APPENDIX - I

NOMENCLATURE OF MATHEMATICAL SYMBOLS

b	= A geometric constant relating to the packing of the filter grains
C	= Local concentration of particles in the flow
c	= A constant
D, D_{BM}	= Coefficient of Brownian diffusion
d_m	= Diameter of filter media, cm
d_p	= Diameter of suspended particle, cm
f_o	= Porosity of clean filter bed
g	= Acceleration due to gravity, cm/sec^2
H	= Headloss
k	= Boltzman's constant
K_1, K_2, K_3	= Constants
L	= Distance from the inlet face of the filter media, cm
N	= Numerical concentration of particles, number/ml
P	= Streaming potential
P_e	= Peclet number, $V_o d_m / D_{BM}$
T	= Temperature, ^oK
U_o, V_o	= Superficial velocity of filtration, cm/sec
X, Y, Z	= Empirical exponents

α = Collision efficiency factor

β, γ, δ = Positive exponents

Λ = Conductivity

η = Transport efficiency of an individual grain

φ = A constant

λ = Filter coefficient

λ_o = Filter coefficient (clean bed condition)

μ = Viscosity, poise

ρ_f = Density of fluid, g/cm^3

ρ_p = Density of suspended particle, g/cm^3

σ = Specific deposit

σ_u = The ultimate or saturation value of specific deposit

ζ = Zeta potential

ζ_m = Zeta potential of filter media

ζ_p = Zeta potential of suspended particle

APPENDIX - II

TABLE A1

STREAMING POTENTIALS OF 0.5 mm SAND WITH VARYING CONCENTRATIONS OF Na^+ ADDED TO TUBE WELL WATER

Tube Well Water; Λ = 770 µmhos/cm

Flow lpm/sq m	40.7	244.0	258.0	278.5	390.5	516.0
H (cm)	7.0	49.5	55.0	58.5	82.0	110.0
P (mV)	0.0	1.0	1.0	1.0	1.5	2.0

Tube Well Water + 0.38 meq/l Na^+; Λ = 818 µmhos/cm

Flow lpm/sq m	179.0	198.5	272.0	294.0	393.0	397.0
H (cm)	32.7	40.5	50.8	54.0	70.0	71.5
P (mV)	0.75	1.0	1.0	1.0	1.25	1.25

Flow lpm/sq m	447.0	472.5	522.5	655.0
H (cm)	72.0	93.5	104.5	125.5
P (mV)	1.25	1.5	2.0	2.25

Tube Well Water + 0.475 meq/l Na^+; Λ = 830 µmhos/cm

Flow lpm/sq m	208.5	248.0	306.0	351.5	361.5	417.0
H (cm)	39.5	46.0	59.5	68.5	70.0	81.0
P (mV)	-0.5	-0.5	-0.75	-0.75	-1.0	-1.0

H = headloss; P = streaming potential; Λ = conductance

(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A2

STREAMING POTENTIALS OF 0.5 mm SAND WITH VARYING CONCENTRATIONS OF Ca^{++} ADDED TO TUBE WELL WATER

Tube Well Water; Λ = 770 µmhos/cm

Flow lpm/sq m	40.7	244.0	258.0	278.5	390.5	516.0
H (cm)	7.0	49.5	55.0	58.5	82.0	110.0
P (mV)	0.0	1.0	1.0	1.0	1.5	2.0

Tube Well Water + 0.095 meq/l Ca^{++}; Λ = 782.8 µmhos/cm

Flow lpm/sq m	133.0	318.0	338.0	388.0	413.0	517.0
H (cm)	25.3	63.0	67.5	80.0	88.0	105.0
P (mV)	0.75	1.25	1.25	1.25	1.75	2.0

Tube Well Water + 0.190 meq/l Ca^{++}; Λ = 795.6 µmhos/cm

Flow lpm/sq m	145.0	195.0	258.5	324.0	366.0	432.0	510.0
H (cm)	29.6	40.0	55.0	69.0	79.0	93.0	112.5
P (mV)	0.75	0.75	1.0	1.25	1.25	1.5	2.0

Tube Well Water + 0.380 meq/l Ca^{++}; Λ = 821.2 µmhos/cm

Flow lpm/sq m	95.3	181.0	282.0	336.0	468.0	540.0
H (cm)	19.0	37.5	60.0	74.0	104.5	123.5
P (mV)	-0.25	-0.5	-0.75	-1.0	-1.5	-1.5

Tube Well Water + 0.475 meq/l Ca^{++}; Λ = 834.0 µmhos/cm

Flow lpm/sq m	188.5	191.0	272.0	402.0	495.0
H (cm)	39.5	42.5	61.5	92.0	120.0
P (mV)	-0.5	-0.5	-0.75	-1.25	-1.75

H = headloss; P = streaming potential; Λ = conductance

(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A3

STREAMING POTENTIALS OF 0.5 mm SAND WITH VARYING CONCENTRATIONS OF Al^{+++} ADDED TO TUBE WELL WATER

Tube Well Water; Λ = 770 μmhos/cm

Flow lpm/sq m	40.7	244.0	258.0	278.5	390.5	516.0
H (cm)	7.0	49.5	55.0	58.5	82.0	110.0
P (mV)	0.0	1.0	1.0	1.0	1.5	2.0

Tube Well Water + 0.095 meq/l Al^{+++}; Λ = 784 μmhos/cm

Flow lpm/sq m	133.0	153.0	222.5	254.5	258.0	357.5	441.0
H (cm)	38.0	52.0	68.5	70.5	75.0	90.0	117.0
P (mV)	0.5	0.75	1.0	1.25	1.50	1.75	2.0

Tube Well Water + 0.190 meq/l Al^{+++}; Λ = 798 μmhos/cm

Flow lpm/sq m	55.5	101.5	143.0	186.8	272.0
H (cm)	32.0	50.5	73.0	83.5	110.5
P (mV)	0.25	0.5	0.75	1.0	1.25

Tube Well Water + 0.285 meq/l Al^{+++}; Λ = 812 μmhos/cm

Flow lpm/sq m	39.75	79.5	91.4	129.0	185.0
H (cm)	40.0	73.5	75.0	89.5	108.5
P (mV)	0.5	0.75	0.75	1.0	1.0

Tube Well Water + 0.380 meq/l Al^{+++}; Λ = 826 μmhos/cm

Flow lpm/sq m	17.9	34.8	54.6	81.4	123.0
H (cm)	32.0	59.0	75.0	96.0	120.5
P (mV)	-0.25	-0.5	-0.75	-1.0	-1.25

Tube Well Water + 0.475 meq/l Al^{+++}; Λ = 840 μmhos/cm

Flow lpm/sq m	61.5	89.4	161.0	171.0	298.0	467.0
H (cm)	33.0	36.0	44.5	84.0	111.0	116.0
P (mV)	-0.5	-0.5	-0.75	-1.0	-1.5	-1.5

H = headloss; P = streaming potential; Λ = conductance
(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A4

STREAMING POTENTIALS OF 1 mm COAL WITH VARYING CONCENTRATIONS OF Na^+ ADDED TO TUBE WELL WATER

Tube Well Water; Λ = 770 µmhos/cm

Flow lpm/sq m	238.5	278.0	285.0	391.5	487.0	635.0
H (cm)	8.0	9.7	10.5	13.5	19.5	25.5
P (mV)	0.25	0.25	0.25	0.25	0.5	0.5

Tube Well Water + 0.19 meq/l Na^+; Λ = 794 µmhos/cm

Flow lpm/sq m	228.5	504.0	497.0	545.0	692.0	834.0
H (cm)	6.5	15.0	15.6	17.0	24.5	32.0
P (mV)	0.0	0.25	0.25	0.25	0.5	0.5

Tube Well Water + 0.38 meq/l Na^+; Λ = 818 µmhos/cm

Flow lpm/sq m	266.0	292.0	437.0	445.0	481.0	586.0
H (cm)	7.5	8.4	13.5	13.8	15.5	20.0
P (mV)	0.25	0.25	0.5	0.5	0.5	0.5

Tube Well Water + 0.475 meq/l Na^+; Λ = 830 µmhos/cm

Flow lpm/sq m	270.0	286.0	445.0	532.5	536.0	544.0	580.0
H (cm)	7.8	8.1	14.7	18.2	18.3	19.5	21.0
P (mV)	-0.25	-0.25	-0.25	-0.50	-0.50	-0.50	-0.5

H = headloss; P = streaming potential; Λ = conductance

(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A5

STREAMING POTENTIALS OF 1 mm COAL WITH VARYING CONCENTRATIONS OF Ca^{++} ADDED TO TUBE WELL WATER

Tube Well Water; Λ = 770.0 µmhos/cm

Flow lpm/sq m	238.5	278.0	285.0	391.5	487.0	635.0
H (cm)	8.0	9.7	10.5	13.5	19.5	25.5
P (mV)	0.25	0.25	0.25	0.25	0.5	0.5

Tube Well Water + 0.095 meq/l Ca^{++}; Λ = 782.8 µmhos/cm

Flow lpm/sq m	121.0	240.0	365.5	582.0	586.0
H (cm)	3.2	7.4	12.0	19.5	20.2
P (mV)	0.0	0.25	0.25	0.5	0.5

Tube Well Water + 0.190 meq/l Ca^{++}; Λ = 795.6 µmhos/cm

Flow lpm/sq m	382.0	407.0	512.0	592.0
H (cm)	11.0	15.5	21.5	24.2
P (mV)	0.25	0.25	0.5	0.5

Tube Well Water + 0.285 meq/l Ca^{++}; Λ = 808.4 µmhos/cm

Flow lpm/sq m	159.0	278.0	286.0	328.0	413.0
H (cm)	5.0	9.9	10.1	11.5	15.2
P (mV)	0.0	-0.25	-0.25	-0.25	-0.5

Flow lpm/sq m	425.0	461.0	715.0
H (cm)	16.0	18.0	31.0
P (mV)	-0.5	-0.5	-1.0

Tube Well Water + 0.475 meq/l Ca^{++}; Λ = 834.0 µmhos/cm

Flow lpm/sq m	155.0	177.0	266.0	328.0	336.0	487.0	596.0
H (cm)	5.5	6.4	10.0	12.8	14.0	20.3	26.5
P (mV)	-0.25	-0.25	-0.50	-0.50	-0.50	-0.75	-1.0

H = headloss; P = streaming potential; Λ = conductance

(81.4 lpm/sq m = 2 gpm/sq ft).

TABLE A6

STREAMING POTENTIALS OF 1 mm COAL WITH VARYING CONCENTRATIONS OF Al^{+++} ADDED TO TUBE WELL WATER

Tube Well Water; Λ = 770 μmhos/cm						
Flow lpm/sq m	238.5	278.0	285.0	391.5	487.0	635.0
H (cm)	8.0	9.7	10.5	13.5	19.5	25.5
P (mV)	0.25	0.25	0.25	0.25	0.5	0.5
Tube Well Water + 0.095 meq/l Al^{+++}., Λ = 784 μmhos/cm						
Flow lpm/sq m	202.0	282.0	288.0	308.0	375.0	715.0
H (cm)	7.2	9.0	10.0	10.5	12.6	26.3
P (mV)	0.0	0.25	0.25	0.25	0.25	0.5
Tube Well Water + 0.190 meq/l Al^{+++}; Λ = 798 μmhos/cm						
Flow lpm/sq m	133.0	230.0	362.0	393.0	606.0	
H (cm)	5.6	9.9	15.5	15.9	25.5	
P (mV)	0.0	0.25	0.5	0.5	0.5	
Tube Well Water + 0.380 meq/l Al^{+++}; Λ = 826 μmhos/cm						
Flow lpm/sq m	105.0	213.0	371.0	526.0		
H (cm)	7.0	15.1	21.3	29.0		
P (mV)	-0.25	-0.25	-0.5	-0.75		
Tube Well Water + 0.475 meq/l Al^{+++}; Λ = 840 μmhos/cm						
Flow lpm/sq m	192.5	218.5	273.0	347.5	357.0	560.0
H (cm)	5.1	5.8	8.1	10.0	11.2	17.5
P (mV)	-0.25	-0.25	-0.5	-0.5	-0.5	-1.0

H = headloss; P = streaming potential; Λ = conductance

(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A7

STREAMING POTENTIALS OF 0.5 mm SAND AFTER ALUM OR POLYELECTROLYTE TREATMENT

Alum Floc Treated

Flow lpm/sq m	157.0	169.0	179.0	244.0
H (cm)	24.0	26.6	28.3	39.8
P (mV)	-0.5	-0.5	-0.5	-0.75

Polyelectrolyte C-41 Treated

Flow lpm/sq m	103.3	109.0	181.0	187.0	195.0
H (cm)	16.3	17.5	30.0	32.8	34.3
P (mV)	-0.25	-0.25	-0.5	-0.5	-0.5

Polyelectrolyte C-41 Treated Filter After Backwashing (15 min at 50 Percent Expansion)

Flow lpm/sq m	393.5	397.5
H (cm)	30.0	30.2
P (mV)	+0.5	+0.5

Polyelectrolyte A-23 Treated

Flow lpm/sq m		248.0		421.0
H (cm)	16.9	32.0	54.5	58.6
P (mV)	+0.25	+0.5	+1.0	+1.0

H = headloss; P = streaming potential

(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A8

STREAMING POTENTIALS OF 1 mm COAL AFTER ALUM OR POLYELECTROLYTE TREATMENT

Alum Floc Treated

Flow lpm/sq m	328.0	341.5
H (cm)	8.3	8.4
P (mV)	-0.25	-0.25

Polyelectrolyte C-41 Treated

Flow lpm/sq m	389.5	413.0	488.0	692.0
H (cm)	11.1	12.2	15.2	23.5
P (mV)	-0.25	-0.25	-0.25	-0.5

Polyelectrolyte A-23 Treated

Flow lpm/sq m	103.3	167.0	179.0	179.0
H (cm)	15.4	27.5	28.8	29.3
P (mV)	0.5	1.0	1.0	1.0

H = headloss; P = streaming potential

(81.4 lpm/sq m = 2 gpm/sq ft)

TABLE A9

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm SAND AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8.0; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	6.230			
1.0	-	5.950	5.300	4.815
2.0	6.050	4.700	-	3.625
3.0	-	6.015	-	3.750
4.0	-	5.815	-	4.350
5.0	4.900	5.700	-	4.450
6.0	6.200	6.125	-	4.375

TABLE A10

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm COAL AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8.0; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		5.10 cm (2.0 in.)	20.35 cm (8.0 in.)	43.20 cm (17.0 in.)
0	6.500			
1.0	-	4.850	3.175	2.500
2.0	4.900	4.450	2.485	2.175
3.0	-	4.225	2.625	2.225
4.0	-	4.050	3.125	2.115
6.0	4.930	4.515	3.600	2.900
8.0	5.630	4.735	3.675	2.325

TABLE A11

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 1 mm COAL AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		5.10 cm (2.0 in.)	20.35 cm (8.0 in.)	43.20 cm (17.0 in.)
0	5.900			
1.0	-	5.665	4.850	4.085
2.0	5.550	5.550	4.535	3.700
3.0	-	5.425	4.400	4.100
4.0	5.770	5.550	4.200	3.800
6.0	-	5.100	4.995	4.675
8.0	5.900	5.415	5.350	4.235

TABLE A12

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm SAND (ALUM FLOC TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.885			
1.0	-	4.050	2.650	1.660
2.0	-	3.865	2.390	1.620
3.0	-	3.875	2.690	2.285
4.0	4.500	4.385	3.497	2.700
6.0	5.500	4.475	3.323	2.515

TABLE A13

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm COAL (ALUM FLOC TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	6.900			
1.0	-	4.246	2.150	0.890
2.0	6.425	4.100	2.330	1.163
3.0	-	4.600	3.000	1.770
4.0	6.700	4.880	3.490	1.790
6.0	6.135	-	3.457	2.260
8.0	-	4.886	3.190	2.715

TABLE A14

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 1.0 mm COAL (ALUM FLOC TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	7.688			
1.0	-	6.125	4.574	1.910
2.0	6.725	5.315	4.286	3.480
3.0	-	5.250	5.086	4.334
4.0	6.700	5.315	-	4.846
6.0	7.700	-	5.060	5.080

TABLE A15

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm SAND AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water + 0.475 meq/l Ca^{++};
pH ≈ 8; Λ ≈ 900 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	7.535			
1.0	-	6.665	4.500	4.150
2.0	7.385	5.285	3.760	3.580
3.0	-	5.485	3.920	3.720
4.0	-	3.985	3.840	3.420
6.0	6.300	4.500	3.830	3.540
8.0	-	4.565	3.825	3.320

TABLE A16

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm COAL AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water + 0.475 meq/l Ca^{++};
pH ≈ 8; Λ ≈ 900 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	7.065			
1.0	-	4.525	3.046	1.514
2.0	6.985	4.865	2.960	2.300
3.0	-	3.675	2.680	2.226
4.0	6.625	4.450	3.400	2.234
6.0	-	4.850	3.254	2.300
8.0	5.815	4.950	3.346	2.230

TABLE A17

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm SAND (C-41 TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		5.10 cm (2.0 in.)	20.35 cm (8.0 in.)	43.20 cm (17.0 in.)
0	7.765			
1.0	-	5.550	2.427	2.377
2.0	7.250	4.165	2.977	2.720
3.0	-	4.365	3.520	-
4.0	6.600	3.850	3.435	2.527
6.0	-	3.465	3.300	2.515
8.0	-	3.350	2.997	2.397

TABLE A18

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 1.0 mm COAL (C-41 TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		5.10 cm (2.0 in.)	20.35 cm (8.0 in.)	43.20 cm (17.0 in.)
0	5.185			
1.0	-	5.100	3.580	2.714
2.0	4.885	4.235	-	3.950
3.0	-	4.265	3.120	2.594
4.0	5.065	4.015	3.640	2.600
6.0	-	4.275	3.120	2.654
8.0	4.550	4.350	3.520	2.566

TABLE A19

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm SAND (A-23 TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.400			
1.0	-	4.000	3.420	2.520
2.0	4.835	3.750	2.640	2.612
3.0	-	3.650	2.920	2.612
4.0	5.135	3.035	2.532	2.240
6.0	-	3.250	3.068	2.092

TABLE A20

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH 0.5 mm COAL (A-23 TREATED) AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 840 μmhos/cm; Temp. ≈ 30°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		5.10 cm (2.0 in.)	20.35 cm (8.0 in.)	43.2 cm (17.0 in.)
0	5.800			
1.0	-	4.125	3.165	1.660
2.0	5.475	4.275	3.255	1.791
3.0	-	3.500	3.090	2.430
4.0	-	3.600	3.040	2.210
6.0	4.815	2.735	2.610	2.520
8.0	-	3.135	2.600	2.440

TABLE A21

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm SAND AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 980 μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	6.250			
1.0	-	4.150	3.160	2.320
2.0	5.875	3.950	3.292	2.400
3.0	-	4.650	3.708	2.560
4.0	6.000	3.715	3.680	2.532
6.0	5.565	4.400	3.820	2.600
8.0	5.175	4.315	3.468	2.708

TABLE A22

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
1.0 mm COAL AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 980 μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	7.000			
1.0	-	5.500	3.794	2.534
2.0	6.785	5.500	4.700	3.246
3.0	-	6.250	4.030	3.314
4.0	-	4.650	4.834	3.110
6.0	6.050	5.075	4.000	3.574

TABLE A23

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm SAND AT 9.8 m/hr (4 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 980 μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.533			
1.0	-	4.165	3.480	2.560
2.0	5.450	4.450	3.388	3.068
3.0	-	4.200	3.572	2.400
4.0	4.500	3.785	3.880	2.748
6.0	-	3.265	3.068	2.692

TABLE A24

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
1.0 mm COAL AT 9.8 m/hr (4 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 980 μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	6.150			
1.0	-	5.150	3.820	3.540
2.0	5.385	4.715	3.772	3.572
3.0	-	4.485	3.440	3.452
4.0	4.715	4.500	3.760	3.640
5.0	-	-	4.240	3.668

TABLE A25

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm SAND AT 14.7 m/hr (6 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 980 μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.658			
1.0	-	4.578	3.428	3.140
2.0	5.220	4.500	3.440	3.412
3.0	-	4.470	3.200	3.220
4.0	5.178	4.380	-	3.140

TABLE A26

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
1.0 mm COAL AT 14.7 m/hr (6 gpm/sq ft)

Tube Well Water; pH ≈ 8; Λ ≈ 980 μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.500			
1.0	-	4.470	3.892	3.660
2.0	5.100	-	4.000	3.720
3.0	-	4.362	4.212	3.680
4.0	5.562	4.710	3.720	3.772

TABLE A27

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm SAND AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH = 6.0 (Adjusted With HCl)
$\Lambda \approx$ 1400 μmhos/cm; Temp. $\approx 36^{o}C$

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.895			
1.0	-	4.465	2.914	1.846
2.0	5.635	3.750	3.110	-
3.0	-	-	2.820	2.280
4.0	-	4.425	2.560	1.680
6.0	5.850	4.265	2.850	1.900

TABLE A28

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
1.0 mm COAL AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH = 6.0 (Adjusted With HCl)
$\Lambda \approx$ 1400 μmhos/cm; Temp. $\approx 36^{o}C$

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	6.185			
1.0	-	4.450	2.754	2.000
2.0	4.865	4.350	3.280	2.214
3.0	-	4.150	2.740	2.446
4.0	4.750	4.300	3.140	2.460
6.0	-	-	3.486	2.820

TABLE A29

FILTRATION OF DISCRETE VIRUS PARTICLES THROUGH
0.5 mm SAND AT 4.9 m/hr (2 gpm/sq ft)

Tube Well Water; pH = 7.0 (Adjusted With HCl)
$\Lambda \approx 1225$ μmhos/cm; Temp. ≈ 36°C

Time (hr)	PFU/ml x 10^3			
	Influent	Depth From Top		
		6.35 cm (2.5 in.)	21.60 cm (8.5 in.)	44.40 cm (17.5 in.)
0	5.185			
1.0	-	3.625	2.594	1.600
2.0	4.915	3.785	2.600	1.970
3.0	-	3.415	2.426	1.680
4.0	5.000	3.415	2.640	1.874
6.0	4.850	3.485	2.654	1.950

TABLE A30

FILTRATION OF VIRUS AND CLAY TURBIDITY THROUGH 0.5 mm SAND AT 4.9 m/hr (2 gpm/sq ft)

Temperature ≈ 30°C; Λ ≈ 900 μmhos/cm; pH ≈ 8

Filtration Condition	Time (hr)	PFU/ml x 10^3				Turbidity FTU		Headloss in the Layer (cm)			
		Influent	Depth From Top			Inf	Eff	0–7.62 cm	0–22.85 cm	0–38.10 cm	0–45.70 cm
			7.62 cm (3 in.)	22.85 cm (9 in.)	45.70 cm (18 in.)						
Untreated Media	0	4.925									
	0.5	-	1.400	1.340	1.330	20	0.53	2.9	6.6	11.1	13.0
	1.5	4.750	1.330	1.180	1.150		3.50	3.4	7.2	11.4	14.1
	3.5	4.265	1.080	1.110	1.100		1.40	4.8	8.9	13.4	16.2
	5.5	-	0.770	0.770	0.620		1.40	8.6	12.7	17.5	20.2
C-41 Treated Media	0	6.960									
	0.5	-	1.530	1.290	0.980	21	0.38	3.2	7.5	11.0	14.1
	1.5	6.262	1.297	1.107	1.013		0.65	4.1	8.8	12.8	15.0
	3.5	4.920	0.690	0.690	0.653		0.48	6.2	11.2	15.4	17.6
	5.5	-	0.640	0.600	0.547		0.56	9.7	14.9	19.1	21.4
Untreated Media; Coagulated and Settled Influent	0	40.000				33					
	0.5	27.400	0.757	0.680	0.680		0.075[φ]	4.0[φ]	9.2[φ]	-	18.5[φ]
	1.5	14.800	0.560	0.530	0.447		0.075	4.4	8.9	-	17.1
	3.5	13.650	0.363	0.343	0.303		0.075	7.0	11.8	17.3	20.0
	5.5	8.855	0.400	0.387	0.357		0.075	11.7	16.8	22.3	25.5

[φ]Values of 1 hr sample.

TABLE A31

FILTRATION OF VIRUS AND CLAY TURBIDITY THROUGH DUAL-MEDIA AT 9.8 m/hr (4 gpm/sq ft)

Temperature ≈ $30^{o}C$; Λ ≈ 900 μmhos/cm; pH ≈ 8

Filtration Condition	Time (hr)	PFU/ml x 10^3				Turbidity FTU		Headloss in the Layer (cm)			
		Influent	Depth From Top			Inf	Eff	0–7.62 cm	0–22.85 cm	0–38.10 cm	0–45.70 cm
			7.62 cm (3 in.)	22.85 cm (9 in.)	45.70 cm (18 in.)						
								Coal		Sand	
Untreated Media	0	6.885									
	0.5	6.735	3.300	3.000	1.940		3.5	1.2	2.6	11.5	15.5
	1.5	6.185	3.206	2.840	2.460	22	3.0	1.5	3.2	14.9	19.5
	3.5	5.975	-	2.510	1.680		2.3	1.7	3.6	16.5	21.0
	5.5	-	-	2.720	1.903		2.3	2.3	4.6	20.0	24.4
C-41 Treated Media	0	6.030									
	0.5	5.370	2.507	2.307	1.550		5.00	1.4	3.6	13.1	16.5
	1.5	-	2.125	2.013	1.383	22	4.00	1.6	4.2	14.9	18.6
	3.5	5.058	2.057	1.230	0.977		3.33	2.5	5.6	18.7	22.4
	5.5	-	1.780	1.145	0.815		2.30	3.7	7.6	23.5	27.4
Untreated Media; Coagulated and Settled Influent	0	28.520				44					
	0.5	24.120	4.930	4.200	1.780	44	0.36	1.3	3.0	13.1	17.3
	1.5	18.750	3.170	2.565	1.530	37	0.36	2.0	4.2	16.1	20.7
	3.5	13.400	2.142	1.870	1.023	33	0.50	3.4	6.2	19.3	23.8
	5.5	8.900	2.045	1.220	0.740	20	0.50	4.7	8.0	22.6	27.3

VITA

Name:	N. Sriramulu
Date of Birth:	June 1, 1938
Education:	School Final, 1955, Nehru Board Higher Secondary School, Nidubrolu, Guntur, Andhra Pradesh.
	Intermediate in Science, 1957, Andhra Loyola College, Vijayawada (Andhra University).
	Bachelor of Civil Engineering, 1961, College of Engineering, Kakinada (Andhra University).
	Master of Science (Engineering), Public Health Engineering, 1963, Annamalai University.
	Candidate for Ph.D. in Civil Engineering, 1972-present, Indian Institute of Technology, Kanpur.
Professional Experience:	Junior Engineer, Public Works Department, Govt. of Andhra Pradesh, 1961-62.
	Associate Lecturer in Civil Engineering, College of Engineering, Osmania University, 1963-64.
	Lecturer in Civil Engineering, Department of Technical Education, Govt. of Andhra Pradesh, 1964-present.
Publications:	'Preliminary Investigation on the Use of Coal for Removing Viruses from Water.' Co-authors: Oza, P.P., Chaudhuri, M. <u>Proc. Symposium on Environmental Pollution</u>, Central Public Health Engineering Research Institute, Nagpur, India (1973).

Discussion on 'Desorption and Retentivity of Filtering Materials' by A. Leclerc and T.T. Vu. Co-author: M. Chaudhuri. J. Env. Eng. Div., Proc. Am. Soc. Civil Engrs., 100, EE1, p. 219 (1974).

Discussion on 'Determing Specific Deposites' by Backwash Technique by K.Y. Hsiung. Co-authors: Ranade, S.V., Chaudhuri, M. J. Env. Eng. Div., Proc. Am. Soc. Civil Engrs. (To appear in April, 1975).